IN THE UNITED STATES DISTRICT COURT FOR THE
NORTHERN DISTRICT OF OKLAHOMA

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W. A. DREW EDMONDSON, in his )
capacity as ATTORNEY GENERAL )
OF THE STATE OF OKLAHOMA and )
OKLAHOMA SECRETARY OF THE    )
ENVIRONMENT C. MILES TOLBERT,)
in his capacity as the       )
TRUSTEE FOR NATURAL RESOURCES)
FOR THE STATE OF OKLAHOMA,   )
                             )
            Plaintiff,       )
                             )
vs.                          )4:05-CV-00329-TCK-SAJ
                             )
TYSON FOODS, INC., et al,    )
                             )
            Defendants.      )
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THE VIDEOTAPED DEPOSITION OF BRIAN HAGGARD PhD, produced as a witness on behalf of the Plaintiff in the above styled and numbered cause, taken on the 16th day of April, 2009, in the City of Fayetteville, County of Washington, State of Arkansas, before me, Lisa A. Steinmeyer, a Certified Shorthand Reporter, duly certified under and by virtue of the laws of the State of Oklahoma.

VIDEOGRAPHER: We are now off the Record. The time is 8:35 a.m.

(Whereupon, a discussion was held off the Record.)

VIDEOGRAPHER: We are back on the Record. The time is 8:38 a.m. 08:39AM

MR. GARREN: Mr. Varady, you wanted to make a Record before we proceeded. Why don't you go ahead and do that now.

MR. VARADY: Thank you, Mr. Garren. I appreciate the opportunity to do that at the outset of the deposition to try to minimize any interruptions with your interrogation. 08:39AM

I'm here today on behalf of Dr. Haggard, who is here as a fact witness in response to a subpoena that you issued. Even though the subpoena did not compel the production of any records from Mr. Haggard, he did bring his resumT at your request. He is prepared to answer factual questions regarding his resumT, his prior and current research and publications and for facts over which he has direct knowledge. 08:39AM 08:40AM

Dr. Haggard is not here today as an expert witness. The Federal Rules of Civil Procedure make clear that he's not legally obligated to render his 08:40AM professional opinions or extrapolate from his research and publications to respond to hypothetical questions or questions pertaining to the ultimate issues in this case. He has not been retained as an expert in this case by either party, not disclosed 08:40AM as an expert witness in any discovery. Dr. Haggard has not read the complaint or conducted any research on the ultimate issues in dispute in this case. No scientific basis exists for Dr. Haggard to render scientific opinions on the ultimate issues in this 08:40AM case. The parties' respective experts can interpret Dr. Haggard's research and publications with regard to their expert opinions.

Federal Rule of Civil Procedure 45(c)3(b)(ii) makes clear that Dr. Haggard can't be compelled to 08:40AM render his professional opinion as an expert for the parties in this case. As stated in the advisory committee note to Rule 45(c)3(b)(ii), quote, the compulsion to testify can be regarded as the taking of intellectual property. The Rule establishes the 08:41AM right of such persons to withhold their expertise, at least unless the parties seeking it makes the kind of showing required for conditional denial of a motion to quash as provided in the final paragraph of 45(c)(b)(iii). That requirement is the same as 08:41AM that necessary to secure work product under Rule 26(b)3 and gives assurances of reasonable compensations, closed quote.

For these reasons and based on the caselaw cited in the advisory committee note, Dr. Haggard 08:41AM objects to any questions or lines of questioning seeking his professional opinion on the issues in this case under rule 45(c)3(b)(ii) and hereby moves to limit the scope of any inquiry to his knowledge of the facts relevant to this case rather than 08:41AM provide opinion testimony or his previously formed opinions as expressed in his publications.

If plaintiff's counsel fails to comply with the requirements of the Rule, Dr. Haggard moves to suspend the deposition to submit a motion to the 08:42AM U.S. District Court for the Western District of Arkansas.

Thank you, Mr. Garren.

MR. GARREN: Is Dr. Haggard willing to accept reasonable compensation today for his 08:42AM testimony?

MR. VARADY: Well, not as an expert witness.

MR. GARREN: Is he willing to accept compensation today for his testimony today? 08:42AM

A Brian Edward Haggard.
Q Okay. Let's hand you what's been marked as Exhibit 7 and tell the court, what that is, if you would, please.
A This is a copy of my resumT or curriculum vitae. 08:44AM
Q Okay. How current is this?
A It was last updated in January of this year I believe.
Q Okay. Let's talk a little bit about your education. You obtained a BS degree at the University of Missouri in Rolla; is that correct? 08:44AM
A Yes, sir.
Q And your major there was life sciences?
A Yes, sir. 08:44AM
Q All right. Next you obtained your masters at University of Arkansas in 1997 in environmental soil and water science. Who was on your committee for your masters thesis?
A Dr. Phillip Moore was my advisor. Dr. Tommy Daniel was a departmental committee member. Dr. Chuck West was a departmental committee member, and I believe Dr. Thoma was the external member. 08:44AM
Q And Thoma, is that T-O-M-A?
A T-H-O-M-A. 08:45AM

Q Thank you.
A Chemical engineering department.
Q Then you obtained your doctorate at Oklahoma State University in biosystem engineering in the year 2000. Tell us, if you would, who was your advisors on your thesis there. 08:45AM
A Dr. Dan Storm was my dissertation advisor.
Q On the committee, who were they?
A Dr. Mike Smolen, Dr. Tom Honn and Dr. Emily Stanley. 08:45AM
Q As of January 1 when your resumT or curriculum vitae was prepared, is it accurate and complete as far as you know?
A I believe so. There might -- there could be some grants that are left off because I haven't updated it. 08:46AM
Q Okay, but at the time it's pretty much complete as far as you can tell?
A Yes, sir.
Q All right. Let's talk about your employment history. Why don't we start when you first had what you considered to be a real job either in college or after college. 08:46AM
A My first real job was working for M&M Environmental Consulting in Fort Smith, Arkansas, 08:46AM

the summer after I finished my bachelors degree.
Q What kind of work did you do there?
A Mostly stabilization of chemicals for their disposal into municipal landfills.
Q How long did that last? 08:46AM
A Two months.
Q And what time frame were you talking about that that occurred?
A That would be June through approximately August 1994. 08:47AM
Q Okay. What would be your next employment after that?
A At that point I went to graduate school, and I did not hold a real job until I was hired by the U. S. Geological Survey in January of 2000. 08:47AM
Q All right. Is that shown then on your curriculum vitae, Exhibit 7, as the hydrologist in the Tulsa office in 2000 to 2002?
A Yes, sir.
Q Okay, and what were your responsibilities there as a hydrologist? 08:47AM
A Water quality data analysis.
Q And how long did that position last?
A I was with the U. S. Geological Survey from 2000 through about August 2001. 08:47AM

Q Starting when in 2000?
A January 2000.
Q Okay.
A Through August 2001. 08:47AM
Q Okay. Your next position then was as a research hydrologist with the USDA; is that correct?
A That's correct.
Q And that was through 2004 based on your Exhibit 7?
A I have that listed out how I progressed 08:48AM through the federal grades. So I was employed by the USDA from August 2001 through January 2006.
Q Okay. Tell us a little bit about what your job responsibilities were with the USDA.
A The main general focus was tackling water 08:48AM quality issues in northwest Arkansas as related to the poultry industry.
Q And what were your duties or responsibilities in that regard?
A To conduct scientific studies, to evaluate the 08:48AM effects of land use on chemical concentrations in streams.
Q Okay. As part of that work, were you required to publish your findings from the research you performed? 08:49AM

A Yes, sir.
Q And you did do so?
A Yes, sir.
Q Okay. Are those publications listed on your curriculum vitae? 08:49AM
A Yes, sir.
Q Was your area of study then limited to the northwest Arkansas area at that time?
A By proximity.
Q What were the elements of concern or constituents of concern that you were researching or studying? 08:49AM
MR. BURNS: Object to form.
A Could you rephrase the question, please?
Q Yeah. What kind of chemicals that might impact land uses were you concerned with or studying? 08:49AM
MR. BURNS: Object to the form.
A Again, could you simplify that?
Q Okay. Let's talk a little bit about what you did maybe. 08:49AM
A Yes, sir.
Q Did you do field research and sampling and sample collection?
A I did some plot studies. The majority of my 08:49AM

work was sampling in streams.
Q Okay. When you sampled in streams, what were you sampling for?
A We focused on nitrogen and phosphorus concentrations. 08:50AM
Q Okay. When you say we, who do you mean by we?
A Me and my research staff, the people that worked for me --
Q And how many was that?
A -- I've had -- 08:50AM
Q Roughly?
A -- on average two to three people per year, one full-time research associate.
Q After your position at USDA, you became an associate professor at the University of Arkansas; 08:50AM is that correct?
A Yes, sir.
Q And you still hold that title there today?
A Yes, sir.
Q Okay, and as I understand it, you are now the 08:50AM director at the Arkansas Water Resource Center; correct?
A As of July last year, yes, sir.
Q Good. All right. Were you an interim director before that? 08:50AM

Q -- it says, it is not surprising that despite not having received litter additions for at least four years previously, the unamended control still had measurable P in the runoff. Why is that is not surprising; can you tell what that means? 09:15AM

A There is phosphorus stored within the soil that can be lost into runoff waters.

Q All right. It then goes on to say, thus, it is apparent that runoff water quality can be affected and eutrophication of surface waters could 09:15AM still potentially occur years after cessation of broiler litter applications. Again, what does that mean in layman's terms, if you would, please?

A Because of the phosphorus that can be stored in soils, you can still have increased phosphorus 09:16AM concentrations in the runoff water.

Q All right. Based on your knowledge, experience, research and review of published literature, do you have an opinion whether or not the -- that some or all of the nutrients and trace 09:16AM metals as were found in the poultry waste in this study, found in the setting -- let me ask it this way differently: The phosphorus concentrations and metals that are described in this report that run off the studied plots, do those chemicals reach 09:17AM streams and waters in the topography or geology that you find typically as used in this study?

MR. BURNS: Object to form.

MR. VARADY: I'm going to renew my objection at the outset that that's requesting a new opinion not contained in the report. To the extent you can understand the question or feel qualified to response to it, go ahead. 09:17AM

A Specifically to these plots, most all the runoff water was collected in the sampling bottles. 09:17AM

Q And I understand that. If these plots don't have a collection bottle at the end to catch that runoff, where does that runoff typically go in real life in landscape?

MR. BURNS: Object to form. 09:17AM

MR. VARADY: I'm going to renew my objections to the extent it asks for formation of a new opinion.

A Runoff is going to move down slope.

Q And other than moving down slope, what happens to it? 09:18AM

A It is very dependent upon what features are down slope of that particular area.

Q Would infiltration be one option about what could happen to that runoff? 09:18AM

A Yes, sir.
Q Evapotranspiration, would that be another possibility?
A Yes, sir.
Q And becoming just surface water runoff would be a third; is that a fair statement? 09:18AM
A Yes, sir.
Q Are there any other processes that you're aware of from your research and studies what would happen to runoff from a field? 09:18AM
A Just keeping in context with the hydrologic cycle, there is storage of water on the field as well.
Q So there might be some ponding?
A Yes, sir. 09:18AM
Q Does that also possibly lead to further infiltration or leaching, or can it?
A It's possible.
Q Okay. Let's look at Page 1010 and in the lower right-hand corner under Flow-Weighted Mean Concentrations title there's a sentence or two there that says, except for P, which is phosphorus, elevated concentrations of plant nutrients in runoff are generally not considered environmentally or ecologically harmful. However, trace metals in 09:19AM 09:19AM

further monitoring. Again, was that a true
statement at the time this was written?
A Yes.
Q All right, and are there -- did you measure
other -- let me back up. Was leaching measured in 09:43AM
this particular study or was it the kind of the
spinoff study by Pirani?
A It was the study by Pirani.
Q Okay. So I'd have to look at that to find
what was measured at this time but reported 09:43AM
differently in a different paper?
A Yes, sir.
Q Okay. What -- what further monitoring is it
you're saying needs to be done? When you say
requires further monitoring, what is it you expect 09:44AM
that to be?
A The intent of that sentence is to say that we
do need to keep monitoring leaching losses.
Q Okay. The phosphorus buildup in the soil that
you've commented on in this statement, in particular 09:44AM
that came off the control field, is -- can that
occur from just over application of poultry litter?
MR. BURNS: Object to form.
MR. VARADY: I'm going to object as well.
A The application of any source of nutrients can 09:45AM

lead to the buildup of those within the soil.
Q So it's not just peculiar to phosphorus; is that what you're telling me?
A Or to any individual source. Commercial fertilizer would behave similarly as well. 09:45AM
Q Does the phosphorus buildup occur because the plant or the crop is unable to uptake it and use it in its growing process?
MR. BURNS: Object to form.
A It does occur because it is applied in excess of the plants' needs. 09:45AM
Q Okay. The thesis -- I assume there's a thesis that came as a result of Menjoulet's work in this case?
A Yes, sir. 09:46AM
Q Did you sign that thesis?
A Yes, sir.
Q Okay. What does it mean when you sign a thesis?
A That that student has successfully defended her research. 09:46AM
Q And when you say successfully defended, who is she defending against?
A She is defending against questions from her masters committee. 09:46AM

from this hill slope.
Q Is that the purpose, though, when you comment about this is a good methodology to help in BMP implementation; was that the goal in a sense?
A The goal of this was -- this was funded through the USDA NRI program, and it was to demonstrate that the methodology can work in northwest Arkansas, and to show that it could be applied to other fields to delineate what's -- where saturation excess occurs or where infiltration excess occurs.
Q Okay. This may be a question of the obvious, but to do that at a different location, in your opinion, does it require all that instrumentation to be set up as was done in this study?
A It does not have to be as high density a setup.
Q As was used in this study?
A As was used in this study.
Q Okay. Let's look at the first page again of your paper, and the next to the last or the last paragraph in the left-hand column, the second sentence, it says, for example, storm runoff plays a major role in phosphorus transport, and diffuse phosphorus pollution is a major contributor to

09:59AM
09:59AM
09:59AM
09:59AM
10:00AM

freshwater systems. Is that a true statement when it was written for this report?
A Yes, sir.
Q Is there anything that's occurred since the writing of that statement that would change your opinion about the truth of that statement? 10:00AM
A No. Storm runoff does play a role in phosphorus transport.
Q The very first sentence under introduction, it says, storm runoff generation is a non-linear process that has surface and subsurface components. What does that mean to a layman that it's a non-linear process, not what it means to a layman, but how can you explain it to a layman? 10:00AM
A To understand the intent of that sentence, it would be best to ask the primary authors, either Mansoor Leh or Dr. Chaubey. 10:01AM
Q Okay. That's fine. I need to ask you about another word in the report, and these aren't numbered, but it's actually the sixth page of the document I believe. 10:01AM
A Okay.
Q And right here where it says, at Savoy, as in numerous Karst areas elsewhere, characterized by heterogeneous and anisotropic pathways that 10:02AM

which is the upper left-hand column, second or third -- about six lines down. It says, this investigation emphasized the need to carefully manage poultry litter because small losses of nutrients compared to the total amount of nutrients produced in a basin may still impact stream nutrient concentrations and export. Was that statement true when you made it then? 10:24AM
A Yes, sir.
Q Do you believe that statement to be true today? 10:24AM
A Yes, sir.
Q Is there anything that's occurred that would cause you to change this opinion as expressed in this study? 10:24AM
A Not that I'm aware of.
Q Okay. Have you had any experience, sir -- I'm going to change subjects on you -- to study in the Eucha-Spavinaw Lake watershed?
A Yes, sir. 10:24AM
Q Okay. When did you have experience working and studying in the Eucha-Spavinaw watershed?
A The Tulsa Metropolitan Utility Authority and the City of Tulsa funded my dissertation research when I was at Oklahoma State University. 10:25AM

Runoff Water Quality from Broiler Litter-Amended Tall Fescue in Response to Natural Precipitation in the Ozark Highlands

B. C. Menjoulet, K. R. Brye,* A. L. Pirani, B. E. Haggard, and E. E. Gbur University of Arkansas

The Arkansas poultry industry produced more than 1.2 billion broiler chickens (*Gallus gallus domesticus*) and generated approximately 1.3 million Mg of broiler litter in 2002. High transportation costs of relocating broiler litter have led to annual land applications near poultry houses, increasing concern for potential surface water contamination from runoff. The objective of this study was to evaluate the effect of broiler litter application rate on runoff water quality in response to natural precipitation. Six plots (1.5 by 6.0 m), located on a Captina silt loam (finesilty, siliceous, active, mesic Typic Fragiudult), were amended with fresh broiler litter at 0, 5.6, and 11.2 Mg ha^{-1} (control, low, and high litter treatments, respectively) once annually for 4 yr (May 2003 through April 2007). Runoff collected after each runoff-producing event was analyzed for soluble nutrients and metals. Cumulative runoff did not differ among litter treatments over the 4-yr study. At times, flow-weighted mean (FWM) concentrations of As from all litter treatments exceeded the maximum contaminant level for drinking water (0.01 mg As L^{-1}). Four-year FWM Fe concentrations and runoff losses were greater ($P < 0.05$) from the high than from the low litter treatment and unamended control, and the 4-yr FWM P concentration from the low litter treatment (3.0 mg L^{-1}) was greater than that from the unamended control (1.8 mg L^{-1}). Since precipitation is temporally variable, evaluating runoff water quality in response to natural precipitation over several years is key to ascertaining the long-term impacts of surface-applied soil amendments like broiler litter.

The United States produced over 21 billion kg of broiler chicken meat in 2006, which resulted in nearly $19 billion in revenue for the poultry industry (USDA-NASS, 2007). United States broiler production is concentrated in several areas including parts of Delaware, Maryland, and Virginia (i.e., the Delmarva Peninsula), as well as Alabama, Georgia, Mississippi, and the Ozark Highlands, which encompasses northeast Oklahoma, southwest Missouri, and northwest Arkansas. Between 1997 and 2002, Arkansas was one of the top five broiler-producing states in the United States and produced more than 1.2 billion broiler chickens in 2002 alone (USDA-NASS, 2002). The four most northwestern counties in Arkansas (Benton, Carroll, Madison, and Washington) were responsible for 28% of the total broilers produced in the state in 2002 (USDA-NASS, 2002). Aside from being a lucrative industry, broiler production also produces a tremendous amount of waste material, generally referred to as litter. Based on estimates from the University of Arkansas Cooperative Extension Service (UA-CES), the Arkansas broiler industry generated between 1.3 and 1.9 million Mg of litter in 2006 (UA-CES, 2006), making management of the organic waste a challenge.

Broiler litter, and poultry litter in general, contains numerous essential plant nutrients including phosphorus (P), potassium (K), nitrogen (N), calcium (Ca), magnesium (Mg), zinc (Zn), copper (Cu), iron (Fe), manganese (Mn), and boron (B; Kunkle et al., 1981; Edwards and Daniel, 1992; Pirani et al., 2006). Since broiler litter has been shown to be economically competitive with commercial inorganic fertilizers as a nutrient source (Chapman, 1996), the typical management method for broiler litter is land application as an organic soil amendment (Sims and Wolf, 1994; Edwards and Daniel, 1993; Stephenson et al., 1990). Dry-matter yields of tall fescue [*Lolium arundinaceum* (Schreb.) Darbys.], bermudagrass [*Cynodon dactylon* (L.) Pers.; Pote et al., 2003), and orchardgrass (*Dactylis glomerata* L.; Hileman, 1973) have been shown to increase in response to poultry litter application. However, excessive application rates may result in the accumulation of some elements to toxic levels in plant tissue (Kingery et al., 1993).

Broiler litter also contains trace metals, such as arsenic (As), cadmium (Cd), chromium (Cr), nickel (Ni), and selenium (Se; Kunkle et al., 1981; Gupta and Charles, 1999; Pirani et al., 2006), and hormones, such as estradiol and testosterone (Finlay-Moore et al., 2000). Trace metals are present in litter due to their use as growth promoters, disease preventatives, and egg production and feather enhancers (Gupta et al., 1997). However, once exposure to

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the environment, trace metals contained in broiler litter pose numerous environmental and health concerns.

Transporting litter has been relatively costly due to the bulkiness of the litter itself (Slaton et al., 2004). As a result, broiler litter in many areas has been repeatedly land-applied to pastures, typically near poultry houses. Before 2005, poultry litter application rates in Arkansas, and in many other areas, were based on plant-N requirements, which resulted in excessive P accumulations in soil (Sharpley et al., 1993; Kingery et al., 1994; Mitchell and Tu, 2006). In Arkansas, litter application rates are now based on soil-test-P levels and the P index (DeLaune et al., 2004a). Other nutrients and metals, such as Ca, Mg, K (Mitchell and Tu, 2006; Kingery et al., 1994), As, Cd, Cu, and Mn (Gupta and Charles, 1999), have also been shown to increase in soil after repeated poultry litter applications.

The environmental fate of nutrients and metals applied to soil via poultry litter can be influenced by their solubility, mobility, and soil adsorption capacity. Runoff from sloped land is a common potential fate of nutrients and metals that enter the environment via land application of poultry litter (Haggard et al., 2003). Soil physical properties such as texture (specifically clay content), bulk density, and organic matter can either limit or increase the runoff potential of nutrients and metals. Therefore, concern over the potential impairment of surface water quality from nutrient and metal runoff is increasing, particularly in the Ozark Highlands region of northwest Arkansas and northeast Oklahoma (Haggard et al., 2003).

Maintaining high surface and groundwater quality in the Ozark Highlands is of particular concern due to the karst topography underlying most of the region. Many pastures and other grasslands in the Ozark Highlands that have received repeated applications of broiler litter in the past reside on sloping land. Therefore, runoff potential and the potential impairment of surface waters from sediments and sediment-adsorbed P are already high due to the general surface relief and topography within the region. A significant portion of the Ozark Highlands is also underlain by fractured limestone bedrock, which creates the potential for rapid downward transport of surface water and soil leachate solution to the groundwater. Since northwest Arkansas is one of the most rapidly expanding areas in the United States, it is critical to maintain high quality surface and groundwater as potable water sources.

Due to the inherent landscape characteristics, numerous runoff studies examining the effects of broiler litter applications have been conducted previously in the Ozark Highlands (Edwards and Daniel, 1993, 1994; Shreve et al., 1995; Moore et al., 1998; Sauer et al., 1999; Pote et al., 2003) and in other regions of intense poultry production, such as North Carolina (Tarkalson and Mikkelsen, 2004, 2007) and Georgia (Pierson et al., 2001; Schroeder et al., 2004; Franklin et al., 2006). Rainfall simulation has been and is still widely used to create runoff events from relatively small plots amended with broiler litter for assessment of runoff concentrations and losses. As a result of rainfall simulation studies, concentrations of numerous nutrients and trace metals have been shown to be greater in runoff from poultry-litter-amended soils than in runoff from unamended soils (Sharpley et al., 1992; Edwards and Daniel, 1994; Moore et al., 1998; Pote et al., 2003). However, rainfall simulations commonly portray a worst-case scenario (Hershfield, 1961; Wood et al., 1999) rather than characterizing typical soil and runoff response to natural conditions. Rainfall amounts and intensities used in rainfall simulation studies are generally much greater and stronger than what occurs naturally (Hershfield, 1961). Only a few studies have been conducted evaluating runoff water quality from broiler-litter-amended soil under natural precipitation, most of which are short-term (i.e., ≤2 yr; Vervoort et al., 1998; Wood et al., 1999; Finlay-Moore et al., 2000; Sistani et al., 2006), and to our knowledge, no study of this kind has been conducted in the Ozark Highlands. Furthermore, it is probable that results based on response to natural precipitation vary greatly compared to results from rainfall simulations.

The objective of this study was to evaluate the effects of broiler-litter application rate on macronutrient (nitrate N [NO_3–N], ammonium N [NH_4–N], dissolved P, dissolved organic carbon [DOC], Ca, K, Mg, and Na) and trace metal (As, Cd, Cr, Cu, Fe, Mn, Ni, Se, and Zn) runoff from tall fescue on a Captina silt loam over a 4-yr period in response to natural precipitation. It was hypothesized that runoff concentrations and cumulative losses increase as litter rate increases.

Materials and Methods

Site Description

Research began in February 2003 at the University of Arkansas Agricultural Research and Extension Center in Fayetteville, Arkansas. Six plots (three side-by-side pairs) were selected from among several plots that had previously received organic amendments, mainly broiler litter, and been subjected to rainfall simulation studies (Edwards and Daniel, 1993, 1994; Shreve et al., 1995). Plots were 6.0-m long by 1.5-m wide, with a 5% west–east slope, and were located on a Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiudult; Harper et al., 1969; USDA-NRCS, 2004). Similar Mehlich-3 extractable P and pH within the top 5 cm of soil indicated the plots had received similar treatments before this study and were the basis for plot selection (Pirani et al., 2006). Sand, silt, and clay in the top 10 cm averaged 0.31 kg kg^{-1} (standard error [SE] = 0.01), 0.63 (SE = 0.01), and 0.06 (SE = 0.01), respectively (Pirani, 2005).

Selected plots had been established with tall fescue in 1990 (Edwards and Daniel, 1994) and had not been subsequently managed to prevent infestation of other grasses or weeds. Consequently, the plots were primarily tall fescue (~75% by area), but also included (~20%) a mixture of warm- and cool-season grasses (i.e., dallisgrass [*Paspalum dilatatum* Poir], johnsongrass [*Sorghum halepense* (L.) Pers.], crabgrass [*Digitaria sanguinalis*], foxtail [*Setaria* Spp.], orchardgrass [*Dactylis glomerata* L.], and Kentucky bluegrass [*Poa pratensis*]) and a few (<5%) invasive forbs (i.e., buckhorn plantain [*Plantago lanceolata*], curly dock [*Rumex crispus*], dandelion [*Taraxacum officinale*], and hop clover [*Trifolium dubium*]).

Rustproof metal frames enclosed the plots to prevent run-on. Aluminum collection gutters were installed at the down-slope end of each plot. Gutters were covered with plastic to prevent direct precipitation input and were slightly angled to direct runoff from the plot into a subsurface collection container.

Average annual precipitation from 1971 to 2000 for Fayetteville is 117 cm and average annual air temperature is 14.2°C. The mean annual high and low temperatures for Fayetteville are 20.0 and 8.3°C, respectively (NOAA, 2006).

Treatments and Experimental Design

Three broiler litter application rates (0, 5.6, and 11.2 Mg ha^{-1} dry-mass basis) corresponding to an unamended control, low, and high litter rate, respectively, were arranged in a randomized complete block with two replications of each treatment. The control plots were left as is and received no litter or supplemental inorganic fertilizers. The low and high litter rates were based on the University of Arkansas Extension Service's maximum single annual application and maximum total annual application of poultry litter, respectively, that were in place when this study began (UA-CES, 2002; Pirani et al., 2006). For the sake of year-to-year consistency, the initial litter rates were maintained throughout the study despite the state's conversion from these recommendations to using the P-index (DeLaune et al., 2004a,b) on a site-specific basis for recommending the appropriate poultry litter rate. The minimum replication (n = 2 per treatment) in this study was a result of an initial leaching study (Pirani, 2005) that limited the number of feasible replications due to the expense of installing and maintaining equilibrium-tension lysimeters below each plot (Brye et al., 1999). However, several subsequent studies have been published based on this same experimental design and limited treatment replication (Brye and Pirani, 2006; Pirani et al., 2006, 2007).

Broiler Litter Application and Characterization

Broiler litter was applied once annually on 30 Apr. 2003, 6 May 2004, 4 May 2005, and 5 May 2006. Broiler litter used in this study was generated from six to eight flocks of broilers (varying between 12 and 18 mo old) and contained an approximately even mixture of sawdust (*Pinus* spp.) and rice (*Oryza sativa* L.) hulls as bedding materials (Pirani, 2005). Litter was manually broadcast across the entire plot as evenly as possible.

Before application, chemical characterization of the broiler litter was conducted according to established procedures for animal manure analysis (Peters, 2003). Broiler litter pH and electrical conductivity (EC) were measured potentiometrically using a 1:2 litter-to-water mixture. Litter subsamples were digested in HNO_3, treated with H_2O_2, and analyzed by inductively coupled argon plasma-optical emission spectrometry (ICAP-OES) for extractable Ca, Cu, Fe, K, Mg, Mn, Na, P, S, and Zn. Nitrate–N and NH_4–N were determined using 2 mol L^{-1} potassium chloride extractant and a Skalar San Plus (Black Rock, Victoria, Australia) automated wet chemistry analyzer. Total N and C were determined by high-temperature combustion using a LECO CN-2000 analyzer (LECO Corp., St. Joseph, MI). Total recoverable Al, B, Cr, Ni, As, Cd, and Se were determined by digesting litter subsamples in HNO_3–HCl solution, treating subsamples with H_2O_2, heating, and analyzing by ICAP-OES (USEPA, 1996). Table 1 summarizes the 4-yr mean broiler litter composition and mean amounts of each element/compound applied in the low and high litter treatments during this study.

Runoff Collection and Analyses

Runoff from each plot was collected after every runoff-producing precipitation event. Total runoff volumes were measured, and up to 350 mL per sample were saved for chemical analyses. Two unfiltered 20-mL subsamples from each plot were used to determine runoff EC, pH, and redox potential. The pH and redox measurements were conducted immediately using a combination electrode (Model 313, Corning, Inc., Corning, NY). Electrical conductivity was measured using a Corning conductivity meter (Model 441, Corning, Inc., Corning, NY). The remaining sample was filtered through a 1.6-μm glass fiber filter to remove debris and filtered again through a 0.45-μm filter. A minimum of 40 mL (when available) were saved for subsequent chemical analyses. A portion (up to 175 mL when available) of the subsamples was acidified with concentrated HCl (2 drops per 10 mL), and the other half was left unacidified. All runoff subsamples were stored at 4°C until analyses were completed.

Total dissolved K, Ca, Mg, Na, As, Cd, Se, Ni, Cu, Zn, Cr, Mn, Fe, and P concentrations were determined on acidified runoff subsamples by ICAP-OES (Spectro Modula model, Spectro Analytical Instruments, Fitchburg, MA; Vista-Pro, Varian Inc., Mulgrave, Australia). Ammonium–N on acidified subsamples and NO_3–N on unacidified subsamples were determined using a Skalar San Plus automated wet chemistry analyzer (Skalar Analytical B.V., The Netherlands). Dissolved organic C was determined on unacidified runoff subsamples using a Shimadzu Total Organic Carbon Analyzer (Model TOC-V CSH, Shimadzu Scientific Instruments, Columbia, MD).

Soil Characterization

Two days before the initial broiler litter application, four soil-core samples were collected and composited into one sample per plot from the top 10 cm. Soil cores removed from the plots were replaced with cores of similar size taken from immediately adjacent to the plots to minimize preferential flow. Samples were dried at 70°C for 48 h, crushed, and sieved through a 2-mm screen. Soil samples were characterized for extractable and total recoverable minerals, organic matter, pH, EC, and total C and N concentrations. Mehlich-3 extractant (Tucker, 1992) was used for determination of extractable P, K, Ca, Mg, and Na by ICAP spectrometry (CIROS CCD model, Spectro Analytical Instruments, MA). Soil organic matter was determined on sieved soil by weight-loss-on-ignition for 2 h at 360°C. A 2:1 soil-to-water mixture was used to determine soil pH and EC potentiometrically with an electrode.

Precipitation Collection

Precipitation was monitored from February 2003 through May 2007 using a funnel-type collector (Likens et al., 1977) as well as a tipping bucket rain gauge associated with a micrometeorological weather station located at the research site. Rainfall volumes were measured and recorded after every significant precipitation event (Pirani et al., 2006).

Table 1. Four-year mean broiler litter composition and element/compound added in the low (5.6 Mg ha^{-1}) and high (11.2 Mg ha^{-1}) litter treatments. Elemental concentrations are reported on a dry-mass basis. Standard errors are reported in parenthesis.

Litter property	Litter composition	Litter treatment Low	High
		——kg ha^{-1}——	
Moisture (g g^{-1})	0.22 (1.15)	–	–
pH	8.34 (0.17)	–	–
EC† (ds m^{-1})	7.23 (650)	–	–
NO_3–N (mg kg^{-1})	958 (86)	5	11
NH_4–N (mg kg^{-1})	2438 (192)	14	27
Total elements			
C (g kg^{-1})	388 (7)	2171	4341
N (g kg^{-1})	46 (2)	257	514
P (g kg^{-1})	21 (2)	116	231
K (g kg^{-1})	33 (1)	186	372
Ca (g kg^{-1})	35 (5)	196	392
Mg (g kg^{-1})	7 (< 1)	37	74
S (g kg^{-1})	6 (< 1)	36	71
Na (mg kg^{-1})	5774 (143)	32	65
Al (mg kg^{-1})	373 (28)	2	4
Fe (mg kg^{-1})	361 (38)	2	4
Mn (mg kg^{-1})	467 (30)	2.6	5.2
Zn (mg kg^{-1})	453 (25)	2.5	5.1
Cu (mg kg^{-1})	389 (32)	2.2	4.4
B (mg kg^{-1})	54 (6)	0.3	0.6
Ni (mg kg^{-1})	13 (1)	0.07	0.15
Cd (mg kg^{-1})	0.3 (< 0.1)	< 0.01	< 0.01
Cr (mg kg^{-1})	9 (1)	0.05	0.11
As (mg kg^{-1})	19 (2)	0.11	0.21
Se (mg kg^{-1})	5 (1)	0.03	0.06

† EC, electrical conductivity.

Plot Management and Vegetation Assessment

Vegetation in the plots was cut to a height of 9 cm and removed using a bagging lawn mower four to eight times per year (Pirani et al., 2006). Before each cutting, above-ground biomass was sampled by hand clipping from inside a 0.25 m^2 frame at two random locations in each plot. Replicate samples were composited for one sample per plot, dried for approximately 1 wk at 55°C in a forced-draft drier, and weighed for dry matter production.

Data Analysis

Four-year cumulative runoff was calculated by summing runoff (mm) from individual sample dates per plot. Mean 4-yr runoff pH, EC, and redox potential were also calculated for individual plots. Four-year, flow-weighted mean (FWM) concentrations (mg L^{-1}) were determined by dividing the total mass (mg) in the runoff solution per plot by the total runoff volume (L) per plot. Cumulative runoff losses (g ha^{-1}) were determined by dividing the total mass (g) in the runoff solution per plot by the plot area (0.0009 ha).

Based on previous statistical evaluations from the larger study (Brye and Pirani, 2006; Pirani et al., 2006, 2007), analysis of variance (ANOVA) was performed using PROC GLM in SAS 9.1 (SAS Institute Inc., Cary, NC) to determine the effect of broiler-litter application rate on the 4-yr cumulative runoff and runoff losses and the 4-yr mean runoff pH, EC, redox potential, and FWM concentrations of nutrients and metals. An ANOVA was also performed to determine the effect of litter rate on annual above ground dry matter production. When appropriate, treatment means were separated by least significant difference (LSD) at the $P = 0.05$ level (Brye and Pirani, 2006; Pirani et al., 2006, 2007). Pearson correlations were also performed between 4-yr mean DOC concentrations and cumulative losses and those for the other runoff constituents using Mintab version 13.31 (Minitab, Inc., State College, PA).

Since ANOVA treats experimental factors as discrete, non-continuous class variables, linear regression analysis was also preformed using Mintab on the 4-yr total runoff, mean runoff pH, redox potential, EC, FWM concentrations, and cumulative runoff losses to ascertain potential significant trends by treating broiler litter application rate as a continuous numeric variable.

Results and Discussion

Pre-Litter Characteristics

Beginning in February 2003, runoff, leaching, and aboveground dry matter production were monitored for 3 mo before the initial litter application (i.e., May 2003) to evaluate plot uniformity. Pre-litter plot monitoring was necessary to demonstrate that any post-application differences could be attributed to the broiler litter application itself rather than to inherent differences among the plots before the initial application.

Precipitation during the 3-mo period totaled 179 mm, which was 98 mm below the 30-yr average for the months of February, March, and April (Pirani et al., 2006; NOAA, 2006). In response to natural rainfall, cumulative runoff averaged 0.33 (SE = 0.18), 0.58 (SE = 0.04), and 0.50 (SE = 0.09) mm from the control, low, and high litter treatments, respectively, but did not differ significantly among treatments. Runoff EC averaged 0.25 (SE = 0.18), 0.20 (SE = 0.04), and 0.32 (SE = 0.09) dS m^{-1} from the control, low, and high litter-rate treatments, respectively, but did not differ significantly among treatments. Similar to runoff and runoff EC, FWM macronutrient (i.e., NO_3–N, NH_4–N, P, DOC, Ca, K, Mg, and Na) and metal (i.e., As, Cd, Cr, Cu, Fe, Mn, Ni, Se, and Zn) concentrations in the runoff did not differ among pre-assigned litter treatments during the 3 mo before broiler litter application. Drainage and soil leachate concentrations below the 90-cm depth plane during the 3 mo before the initial litter application did not differ among pre-assigned litter treatments (Pirani et al., 2006). Furthermore, total above ground dry matter production during the 3 mo before the initial litter application did not differ among the pre-assigned litter treatments (Pirani et al., 2006). In addition, soil chemical properties (i.e., Al, As, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, S, Se, and Zn) in the top 10 cm, determined from samples collected the day before the initial litter application (Pirani, 2005), did not differ among the pre-assigned litter treatments (Table 2).

Based on the same contention as set forth by Pirani et al. (2006), the lack of pre-assigned litter treatment effects on soil surface chemical properties immediately before the initial litter application and above ground dry matter production, drainage, runoff, and runoff chemistry in the 3 mo before the initial litter application demonstrates the similarity of the plots before this study began. Therefore, it is reasonable to assume that any post-application differences observed in this study can be interpreted as a litter treatment effect rather than being due to major inherent differences among plots before beginning the study.

Vegetative Response

Similar to the results observed by others (Huneycutt et al., 1988; Pote et al., 2003; Brye and Pirani, 2006), broiler litter application increased dry matter (DM) yields of tall fescue. Over the 4-yr study, DM production averaged 5300, 9100, and 11,800 kg ha^{-1} yr^{-1} from the unamended control, low, and high litter treatments, respectively (Table 3). In all 4 yr of the study, DM production from one or both of the litter-amended treatments was greater ($P < 0.05$) than that from the unamended control (Table 3). However, DM production from the high litter was greater ($P < 0.05$) than that from the low litter treatment in only 2 of the 4 yr, while DM production from the low litter treatment was also greater ($P < 0.05$) than that from the unamended control in only 2 of the 4 yr (Table 3). Dry matter production from the low litter treatment was the most variable over the 4-yr study, ranging from 7600 to 11,000 kg ha^{-1} yr^{-1}, while DM production ranged from 4900 to 5600 and 11,300 to 12,200 kg ha^{-1} yr^{-1} from the unamended control and high litter treatments, respectively (Table 3). Though not formally compared, numerically lower DM yields in Year 3 (Table 3) were likely the result of the 37% below-average precipitation that occurred that year (NOAA, 2006; Table 4).

Four-Year Cumulative Effects on Runoff and Runoff Water Quality Runoff

Integrating all individual sample dates over the 4 yr of this study where precipitation varied widely above and below the 30-yr average (Table 4) resulted in a long-term assessment of the effects of broiler litter application rate that more closely represents the natural environmental response compared to rainfall simulation studies. However, 4-yr cumulative runoff was surprisingly small and, in contrast to the hypothesis, did not differ among the broiler litter treatments (Table 5). The 4-yr cumulative runoff measured in this study (6.0 mm) accounted for less than 0.1% of the cumulative precipitation (Table 4; Fig. 1).

The rather small cumulative runoff measured in this study from a 5% slope was somewhat surprising. However, based on the measured particle-size distribution, the Captina silt loam has an estimated surface hydraulic conductivity of 1.4 cm h^{-1} (Saxton et al., 1986), which is a moderately high hydraulic conductivity (USDA-NRCS, 2007) that would tend to allow for infiltration and would transmit water well through the surface layer deeper into the soil profile. Furthermore, for the initial 2 yr of this study — and from the same plots as used in this study — Pirani et al. (2006) reported that drainage below the 0.9-m depth plane accounted for between 21 and 70% of the total annual precipitation depending on litter treatment. Therefore, with the relatively large measured fraction of precipitation exiting the tall fescue root zone via drainage, it stands to reason that infiltration would be high and runoff would be quite low. Had this study been conducted with larger plot areas and in a non-constructed slope and landscape position setting, considerably more runoff would have been expected due to the additional variable-source-area hydrology mechanisms (Loague and Abrams, 2001). Similar to the results of this study, runoff

Table 2. Soil physical and chemical property summary for the 0- to 10-cm depth interval across all plots ($n = 6$) before the first broiler litter application in May 2003. No soil properties differed among pre-assigned litter treatments before the initial litter application.

Soil property	Mean (± standard error)
Bulk density (g cm^{-3})†	1.26 (0.02)
Organic matter (g kg^{-1})‡	31.2 (0.3)
pH †	6.25 (0.04)
Electrical conductivity (dS m^{-1})‡	0.120 (< 0.01)
Mehlich-3 extractable (mg kg^{-1})	
P‡	192 (9)
K‡	193 (2)
Ca‡	1121 (34)
Mg‡	114 (4)
S	12.9 (0.4)
Na‡	12.3 (0.3)
Fe	172 (4)
Mn	146 (6)
Zn	13.2 (0.5)
Cu	4.5 (0.5)
Total recoverable (mg kg^{-1})	
As	5.13 (0.2)
Cd	0.12 (0.02)
Cr	27.6 (1.6)
Ni†	5.5 (0.4)
Se§	< 0.5 (0)
Mg	257 (10)
S	157 (3)
Fe†	15177 (474)
Zn†	41.8 (1.8)
Cu†	11.2 (1.2)
Al	4197 (52)

† Data taken from Pirani (2005).

‡ Data taken from Pirani et al. (2006).

§ Selenium concentrations were below detection in all plots.

Table 3. Broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}) effects on annual above ground dry matter production. Treatment means ($n = 2$) along with coefficients of variation (CV) and least significant difference (LSD) at the 0.05 level are reported.

Litter treatment	Year 1†	Year 2†	Year 3	Year 4
	kg ha^{-1}			
Control	4945a‡	5610a	5334a	5338a
Low	8676ab	9179b	7574a	10,966b
High	11,988b	12,196c	11,371b	11,669b
CV (%)	38	33	36	34
P-value	0.04	0.01	0.04	0.02
$LSD_{0.05}$	4568	2169	3630	3086

† Years represent the following time periods: Year 1, May 2003 – April 2004; Year 2, May 2004 – April 2005; Year 3, May 2005 – April 2006; Year 4, May 2006 – April 2007. Data for years 1 and 2 were taken from Pirani (2005).

‡ Means in the same column followed by different letters are significantly different.

Table 4. Summary of the 4-yr on-site precipitation recorded at the study site and the seasonal and annual 30-yr average precipitation data for Fayetteville, AR.

Time period	Year 1†	Year 2	Year 3	Year 4	4-yr total	30-yr average‡
	mm					
Summer§	334	374	228	151	1087	343
Fall	242	116	192	571	1121	294
Winter	243	384	93	305	1025	256
Spring	331	199	226	161	917	277
Annual	1150	1073	739	1188	4150	1170

† Years represent the following time periods: Year 1, May 2003 – April 2004; Year 2, May 2004 – April 2005; Year 3, May 2005 – April 2006; Year 4, May 2006 – April 2007.

‡ Data taken from NOAA (2006).

§ Seasons are defined as follows: Summer (May, June, and July); Fall (August, September, and October); Winter (November, December, and January); and Spring (February, March, and April).

amounts in response to natural precipitation reported by Wood et al. (1999) also did not differ between litter application rates of 9 and 18 Mg ha^{-1}.

Intensity and duration of rainfall events (Edwards and Daniel, 1993), near-surface soil properties, and antecedent soil moisture conditions have a dramatic impact on the amount of runoff produced from individual precipitation events. Though cumulative runoff did not differ among litter treatments, litter application could affect the number of runoff-producing events that occur due to differences in DM production as a result of the litter's fertilization effect and the subsequent potential effects on soil aggregation (McDowell and Sharpley, 2003), structure, and surface hydraulic conductivity (Franzluebbers, 2002; Rhoton et al., 2002). Therefore, to test this hypothesis, the number of runoff-producing events (i.e., runoff ≥ 0.01 mm, the amount of runoff which produced a sufficient volume of water to generally carry out all desired chemical analyses on individual sample dates) and those events that resulted in only a trace amount of runoff were summed by plot and averaged across years. The number of mean annual trace runoff events were unaffected by litter treatment and averaged 8.1 yr^{-1} across all treatments (Fig. 1). However, the number of runoff-producing events was greater ($P = 0.017$) from the low litter treatment (19.4 yr^{-1}) than from the unamended control (15.2 yr^{-1}) and high litter treatment (12.7 yr^{-1}), which did not differ (Fig. 1). Averaged across all treatments and years, the number of runoff-producing events (i.e., runoff ≥ 0.01 mm) was 15.8 yr^{-1} compared to an average of 44.7 measurable precipitation events per year (data not presented).

Similar to results of rainfall simulations (Edwards and Daniel, 1993), intensity and duration appeared to have a dramatic impact on runoff amounts as demonstrated in Fig. 1. Steep increases in cumulative precipitation represent high-intensity, short-duration rainfall events and were mimicked by rapid increases in cumulative runoff from all treatments (e.g., approximately day of year [DOY] 150 and 103 during Year 1; DOY 300 during Year 4; Fig. 1). Low-intensity and long-duration precipitation events, which mostly occurred during the majority of Year 3 (Fig. 1), had a much smaller cumulative effect on runoff from the three treatments and tended to result in smaller treatment differences.

Table 5. Broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}) effects on the 4-yr cumulative runoff and flow-weighted mean runoff concentrations. Treatment means were separated by least significant difference (LSD) at the 0.05 level when appropriate. Coefficients of variation (CV) are also reported.

Runoff property	Litter treatment Control	Low	High	CV (%)	*P*-value	$LSD_{0.05}$
Runoff (mm)	4.9	7.4	5.6	22	0.17	–
Runoff constituent (mg L^{-1})						
NO_3–N	0.28	0.38	0.68	51	0.22	–
NH_4–N	1.50	2.25	2.10	27	0.34	–
DOC	13.5	19.0	12.7	22	0.15	–
Ca	7.23	5.55	7.43	15	0.20	–
K	13.2	20.2	15.4	25	0.14	–
Mg	1.38	1.66	1.54	11	0.34	–
Na	2.19	2.39	2.38	8	0.71	–
TDP†‡	1.83a	2.98b	2.29ab	24	0.05	0.81
As	0.03	0.04	0.07	60	0.29	–
Cd	< 0.01	< 0.01	< 0.01	24	0.22	–
Cr	< 0.01	< 0.01	< 0.01	18	0.51	–
Cu	0.01	0.01	0.01	16	0.84	–
Fe‡	0.19ab	0.09a	0.30b	51	0.04	0.14
Mn	0.07	0.07	0.05	20	0.22	–
Ni	< 0.01	< 0.01	< 0.01	21	0.19	–
Se	0.01	0.01	0.01	17	0.64	–
Zn	0.14	0.15	0.14	15	0.93	–

† TDP, total dissolved phosphorus.

‡ Means in the same row followed by different letters are significantly different.

Runoff pH, Redox Potential, and Electrical Conductivity

When litter rate was treated by ANOVA as a discrete variable, 4-yr mean runoff pH, redox potential, and EC were unaffected ($P > 0.05$) by litter treatment. Mean annual runoff pH, redox potential, and EC averaged 6.7, 28.3 mV, and 0.36 dS m^{-1}, respectively. This result was somewhat unexpected. It was hypothesized that runoff pH and EC would increase as broiler litter application rate increased due to greater losses of base-forming cations and salts, respectively, as the amount of applied litter increased. However, when treated as a continuous variable, linear regression analysis demonstrated that runoff EC decreased significantly as litter rate increased (Fig. 2). Generally greater DM production, hence greater plant uptake of nutrients, from the high than the low litter treatment and unamended control (Table 3) likely resulted in reduced availability of most soluble macronutrients to be lost via runoff.

Flow-Weighted Mean Runoff Concentrations

Except for P, elevated concentrations of plant nutrients in runoff are generally not considered environmentally or ecologically harmful. However, trace metals in runoff potentially pose a greater environmental threat. Though only two replicate observations per treatment likely increased treatment variability and decreased statistical power to draw significant inferences, 4-yr FWM runoff concentrations of total dissolved P (TDP) and Fe were significantly affected, while 4-yr FWM concentrations of all other plant macronutrients and trace metals evaluated over the course of this study were unaffected by the litter treatments (Table 5). Throughout the 4-yr study, annual

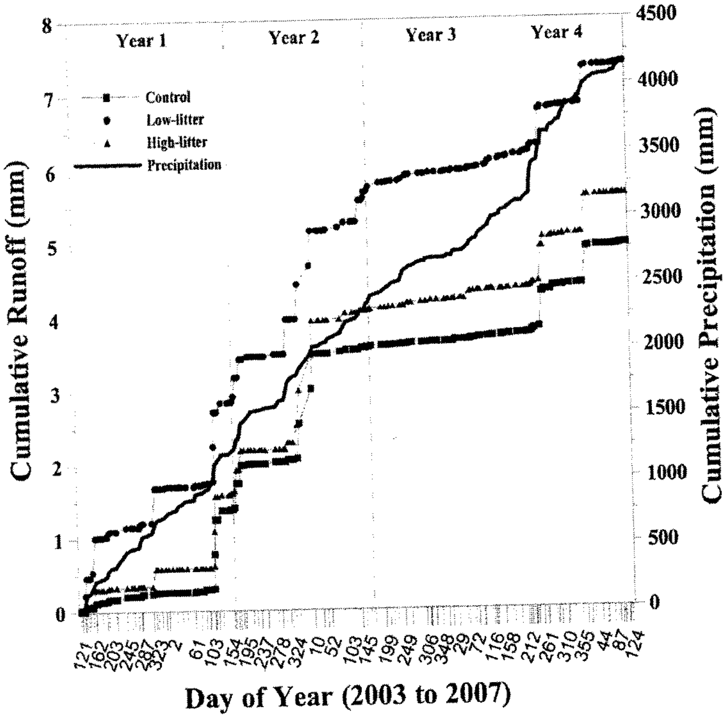

Fig. 1. Mean cumulative runoff (left y-axis) by broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}) and cumulative precipitation (right y-axis) from May 2003 through April 2007.

FWM runoff concentrations of NO_3–N ranged from 0.2 to 3.4 mg L^{-1} yr^{-1}, NH_4–N ranged from 0.8 to 4.6 mg L^{-1} yr^{-1}, Ca ranged from 1.8 to 13 mg L^{-1} yr^{-1}, K ranged from 8 to 73 mg L^{-1} yr^{-1}, Mg ranged from 0.7 to 4.7 mg L^{-1} yr^{-1}, Na ranged from 0.9 to 5 mg L^{-1} yr^{-1}, Mn ranged from 0.02 to 0.36 mg L^{-1} yr^{-1}, and Cu ranged from 0.01 to 0.04 mg L^{-1} yr^{-1} (Menjoulet, 2007). Annual FWM concentrations of Cd, Cr, Ni, and Se were all <0.01 mg L^{-1} yr^{-1} and below detection limits on most individual sample dates.

Four-year FWM TDP concentrations were greater ($P = 0.05$) from the low litter treatment than from the unamended control, whereas 4-yr FWM TDP concentrations from the high litter did not differ from the low litter treatment or unamended control (Table 5). Rapid soil fixation of P from the broiler litter along with increased plant uptake from greater DM production (Table 3) likely reduced P in the runoff from the high compared to the low litter treatment. Similarly, Wood et al. (1999) also observed greater TDP concentrations in runoff as broiler litter application rate increased when applied to corn (*Zea mays* L.) on a silty clay Paleudult in Alabama.

Four-year FWM Fe concentrations were greater ($P = 0.04$) from the high than from the low litter treatment, while the 4-yr FWM Fe concentration from the unamended control was similar to that from both the low and high litter treatments (Table 5). High inputs of organic matter can increase Fe mobility in the soil (Kabata-Pendias, 2001), thereby increasing the potential of Fe loss via runoff. Greater amounts of organic matter and Fe were applied to high litter treatment, which likely contributed to the increased 4-yr FWM Fe concentration in the high litter treatment and supports the original hypothesis.

With twice the litter and nutrient mass applied in the high than in the low litter treatment (Table 1), it was expected that runoff concentrations would be greater from the high than low litter treatment and greater from the low litter treatment than the unamended control. In addition, with twice the mass of organic matter applied in the high than in the low litter treatment, as microorganisms decomposed the organic matter, greater amounts of plant nutrients and trace metals were expected to be mineralized, and potentially run off, in the high than in the low litter treatment. Similarly, Edwards and Daniel (1993) reported increased litter-constituent concentrations in runoff as poultry litter application rate increased. However, in this study, annual FWM TDP, DOC, and Zn were all greater from the low litter than from the other two treatments (Table 5).

When litter rate was treated as a continuous variable, regression analyses demonstrated that none of the 4-yr FWM concentrations of the elements or compounds evaluated in this study increased as litter rate increased. This result is somewhat different from the

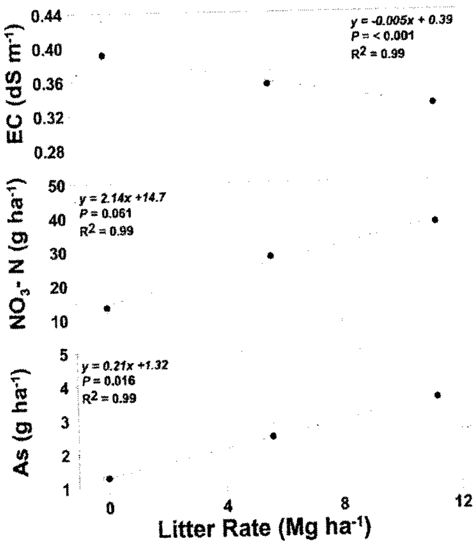

Fig. 2. Linear regression results for broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}) effects on 4-yr mean runoff electrical conductivity (EC) and cumulative runoff losses of NO_3–N and As. Error bars represent standard errors for cumulative mean values (n = 2).

rainfall simulation results of Moore et al. (1998) who reported increased Cu, Zn, and As as litter rate increased. Similar to the results of Moore et al. (1998), annual FWM As concentrations were numerically greater from the high than from the low litter treatment and unamended control, but all FWM As concentrations were less than 0.1 mg As L^{-1}. Arsenic uptake by tall fescue has also been shown to be quite small and was unaffected by broiler litter application rate (Brye and Pirani, 2006). Furthermore, As tends to have similar behavior in soil as P (Jones, 2007), in which P is generally considered immobile and tends to accumulate at the soil surface. Though the unamended control did not receive litter during this study, applications of As-containing poultry litter had been made to the control plots in the past, thus As was present in the top 10 cm of the soil at the beginning of this study (Table 2) to be subjected to possible runoff.

In a rainfall simulation study, Edwards and Daniel (1993) reported that runoff concentrations of soluble littet constituents were generally greatest after the first rainfall event, decreasing with successive simulated rainfall applications, and that runoff losses of nutrients tended to increase as litter application rate increased. Results similar to those of Edwards and Daniel (1993) were rarely demonstrated in this study, but a few occasions did fit this trend, which further indicates how long-term runoff response to natural precipitation is likely much different than the short-term runoff response from rainfall simulation studies that attempt to duplicate extreme rainfall conditions.

Despite relatively few significant litter-treatment effects, FWM concentrations for some elements observed during this 4-yr study were environmentally significant. All annual FWM TDP concentrations from each treatment (Table 5) exceeded the minimum P concentrations of 0.002 to 0.09 mg L^{-1} required for algae growth (Chu, 1943; Sawyer, 1947). Considering the initial Mehlich-3-extractable soil-P concentration in the top 10 cm averaged 192 mg kg^{-1} across all plots before the first litter application (Table 2), it is not surprising that, despite not having received litter additions for at least 4 yr previously, the unamended control still had measurable P in the runoff. Thus, it is apparent that runoff water quality can be affected and eutrophication of surface waters could still potentially occur years after cessation of broiler litter applications. However, with the small amount of runoff measured in this study, the potential for significant ecological impacts would be rather low. Unlike FWM TDP concentrations, FWM Ca concentrations did not meet the minimum requirements of 20 mg L^{-1} needed for algae growth (Walker, 1953; Allen and Arnon, 1955; Wood et al., 1999) throughout this study (Table 5). The karst topography throughout the region in which this study was conducted results in surface water (i.e., runoff water) being hydrologically connected directly to groundwater, which serves as the drinking water supply for many residents. Therefore, drinking water standards are relevant for comparison to runoff concentrations in order to judge the potential environmental threat of runoff from broiler litter-amended soils.

Regulations regarding maximum contaminant levels (MCL) of constituents in drinking water are set by the USEPA, and, currently, NO_3–N is the only plant macronutrient that was measured during this study with a drinking water MCL regulation. Four-year FWM NO_3–N concentrations (Table 5) did not exceed the MCL of 10 mg L^{-1} (Table 6; USEPA, 2006) during this study. Though specific NO_3–N concentrations as a result of individual runoff events were not presented, on six occasions (one from the low litter and five from the high litter treatment) NO_3–N runoff concentrations exceeded the MCL for drinking water.

Flow-weighted mean concentrations of Cd, Cr, Cu, Se, and Zn (Table 5) did not exceed USEPA's recommended MCL for drinking water (Table 6) during this study. However, FWM Fe concentrations from the high litter treatment and unamended control (Table 5) were at times greater than the MCL (Table 6), and, on occasion, FWM Fe concentrations from the unamended control and high litter treatment exceeded the MCL for Fe simultaneously.

The drinking water MCL for As is currently 0.01 mg L^{-1} (Table 6; USEPA, 2006). Annual FWM As concentrations exceeded the As MCL in 3 of 4 yr for all treatments. During the 3-mo span between August and October 2004, the FWM As concentration measured from the unamended control was 33 times greater than the drinking water MCL for As. However, though not measured in this study, the total concentration in runoff water of many elements is generally more environmentally relevant than the soluble or dissolved concentration.

Though runoff water is not directly used for drinking water purposes, much of the runoff that occurs in the Ozark Highlands eventually reaches Beaver Lake, which is the main drinking water source for a significant fraction of the population in rapidly expanding, northwest Arkansas. Therefore, it appears that land application of As-containing broiler litter may pose eventual human and/

Table 6. Summary of selected nutrient and trace metal maximum contaminant levels (MCL) for drinking water and concentration ranges observed in surface and ground water (nutrients) and bed sediments (trace metals) in the Ozark Highlands between 1992 and 1995 (Petersen et al., 1998). Also summarized are other background levels of selected water constituents for various water types.

Element/Compound	MCL†	Concentration range: Surface water	Ground water	Bed sediments
		mg L^{-1}		µg g^{-1}
Dissolved NH_3–N	NR‡	0–0.4	0–0.8	NR
Dissolved NH_3–N + organic N	NR	0.4–1	0.5–0.9	NR
Dissolved NO_3–N + NO_2–N	10/1.0§	0.08–10	0.08–30	NR
Dissolved P	NR	0–1	0–0.5	NR
Zn	5.0	0–0.27	NR	40–5600
As	0.01	NR	NR	3–30
Cd	0.005	NR	NR	0.2–40
Cr	0.1	NR	NR	30–200
Cu	1.3	NR	NR	10–40
Ni	NR	NR	NR	10–40
Se	0.05	NR	NR	0.3–2
Fe	0.3	NR	NR	NR
	Other background levels (mg L^{-1})			
NO_3–N in shallow groundwater¶				2.0
NO_3–N + NO_2–N for EPA Region 5 lakes and reservoirs#				0.01
NO_3–N + NO_2–N for EPA Region 5 rivers and streams*				0.26
Orthophosphate in shallow groundwater ¶				0.02
Total P for EPA Region 5 lakes and reservoirs#				0.033
Total P for EPA Region 5 rivers and streams††				0.068

† Data taken from USEPA (2006).

‡ NR, not reported.

§ MCL for NO_3–N is 10 mg L^{-1}; MCL for NO_2–N is 1.0 mg L^{-1}.

¶ Data taken from USGS (1999).

Data taken from USEPA (2001a). Region 5 includes Arkansas, Louisiana, New Mexico, Oklahoma and Texas.

†† Data taken from USEPA (2001b). Region 5 includes Arkansas, Louisiana, New Mexico, Oklahoma and Texas.

or environmental health risks due to As-enriched runoff. Furthermore, areas that have had a history of repeated annual additions of As-containing poultry litter, but currently do not receive any litter due to high soil-test P, may continue to be an environmental risk for years to follow. However, despite occasional litter treatment differences in FWM concentrations, the rather small amount of runoff measured in this study greatly controlled the mass loss of nutrients via runoff, but also likely resulted in what most would consider rather low or insignificant ecological impacts.

Runoff Losses

Annual runoff losses of NO_3–N, NH_4–N, TDP, DOC, Ca, K, Mg, Na (Fig. 3), Cr, Cu, Mn, Ni, Zn (Fig. 4), and Cd did not differ among litter treatments during any annual period throughout the 4-yr study. Despite the lack of a litter treatment effect on runoff losses of most plant macronutrients, annual runoff losses of Se, Fe, and As differed among litter treatments during one or more years of this 4-yr study (Fig. 4).

Except for Cd and Fe, cumulative 4-yr runoff losses did not differ among litter treatments for any element or compound evaluated in this study (Table 7). Cumulative 4-yr runoff losses of Cd and Fe were the only elements that differed among litter treatments when litter rate was treated as a discrete variable in ANOVA (Table 7). Cumulative Cd runoff losses were two-fold greater ($P = 0.01$) from the low litter treatment than from the unamended control and high litter treatment, which did not differ (Table 7). In contrast, the high litter had greater ($P = 0.03$) 4-yr cumulative Fe runoff losses than the low litter treatment and unamended control, which did not differ (Table 7). Similar to the 4-yr FWM Fe concentrations, runoff losses of Fe from the high litter treatment may have been greater due to increased organic matter, while annual plant uptake remained similar between the high and low litter treatments (Brye and Pirani, 2006), which may have facilitated increased Fe mobility. Furthermore, though Fe is ubiquitous in most soils, particularly in the highly weathered Ultisols of the Ozark Highlands, more readily mobile Fe was likely added via the broiler litter applications in the high litter compared to the low litter treatment.

When litter rate was treated as a continuous variable, regression analyses demonstrated that runoff losses of As increased ($P < 0.016$) as litter rate increased (Fig. 2). Arsenic speciation, thus As mobility, are highly dependent on pH and redox potential (Marera and Le Hécho, 2001). Runoff pH and redox potential were similar throughout the study, suggesting that similar, moderately reduced forms of As (redox potentials of 0 to 100 mV; Matera and Le Hécho, 2001) would be present in runoff, and any differences that would occur would be the result of litter application and/or the rate of application. As was hypothesized, increasing the litter application rate likely resulted in increased As runoff losses. Similar to As, though only marginally significant, runoff losses of NO_3–N also increased ($P = 0.061$) as litter rate increased (Fig. 2). These results demonstrate that increasing rates of broiler litter can have a significant negative impact on the environment.

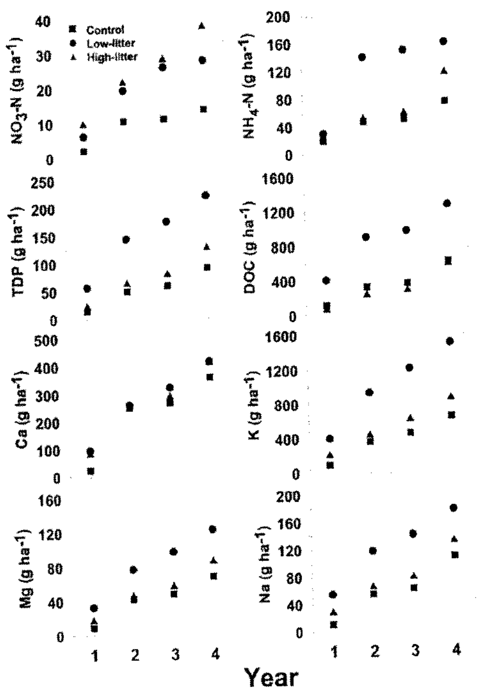

Fig. 3. Cumulative annual runoff losses of macronutrients by broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}).

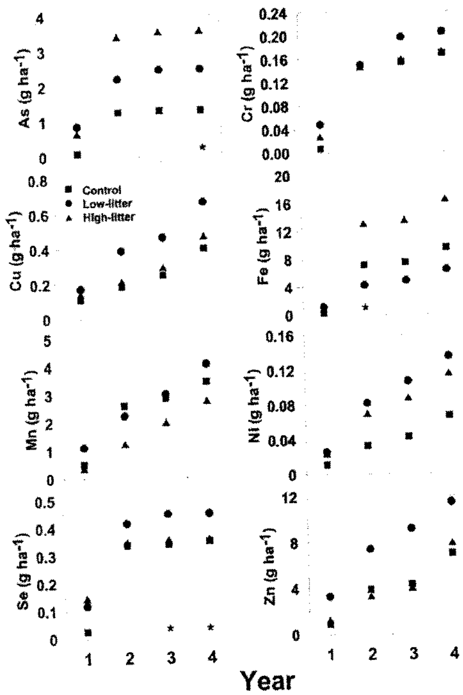

Fig. 4. Cumulative annual runoff losses of trace metals by broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}). Asterisks (*) denote significant treatment difference ($P < 0.05$) in annual runoff losses for that year. Arsenic runoff losses in Year 4 were greater from the high (0.016 g ha^{-1}) than the low litter treatment (0.008 g ha^{-1}) and unamended control (0.005 g ha^{-1}), which did not differ. Iron runoff losses in Year 2 were greater from the high (12.2 g ha^{-1}) than the low litter treatment (3.13 g ha^{-1}) and unamended control (6.91 g ha^{-1}), which did not differ. Selenium runoff losses in Year 3 were greater from the low (0.033 g ha^{-1}) than the high litter treatment (0.009 g ha^{-1}) and unamended control (0.003 g ha^{-1}), which did not differ. Selenium runoff losses in Year 4 were greater from the unamended control (0.01 g ha^{-1}) than the low (0.001 g ha^{-1}) and high litter (0.001 g ha^{-1}) treatments, which did not differ.

Cumulative runoff losses of plant macronutrients (Fig. 3) and trace metals, such as Cu, Mn, Ni, and Zn (Fig. 4), appear to be increasing over time. This result is not surprising for the litter-amended treatments since litter has been applied once a year for 4 yr. However, though generally of lower magnitude, cumulative runoff losses of these same elements/compounds also appear to be increasing from the unamended control. In contrast, cumulative runoff losses of As, Cr, and Se from all treatments appeared to be reaching a plateau by years 3 and 4 (Fig. 4). Although this result suggests that future runoff losses of these trace metals may be negligible despite continued litter application, runoff losses are highly dependent on the timing and intensity of the first several rainfall events after litter application (Edwards and Daniel, 1993; Schroeder et al., 2004).

Runoff losses of DOC were greatest from the low and lowest from the high litter treatment at times (Fig. 3). It is possible that the high litter treatment, even within the first year after litter application, developed a more favorable near-surface environment for soil microorganisms. Tomlinson et al. (2007) reported greater microbial biomass in the top 5 cm of a similar Captina silt-loam soil that had been amended with 9 Mg ha^{-1} of non-alum-treated poultry litter for nine consecutive years compared to a 2.2 Mg ha^{-1} rate and an unamended control. With greater near-surface microbial biomass in response to a high litter rate, the microorganisms present would presumably require a greater amount of readily available C (i.e., DOC) and N to maintain normal function. In this study, DOC runoff losses were at least numerically lower from the high than low litter treatment most of the time (Fig. 3). Furthermore, though NO_3–N and NH_4–N runoff losses were similar among litter treatments during Year 1 (Fig. 3), both NO_3–N and NH_4–N runoff losses were numerically lower from the high than the low litter treatment during Year 1. These observations support the contention that potentially greater microbial biomass under the high than low litter treatment may be effectively utilizing and reducing DOC and inorganic N losses via runoff.

Environmental Significance

Runoff concentrations and losses of some plant nutrients and metals in the Ozark Highlands may be linked to organic matter, specifically C. The loss of dissolved and/or particulate C in runoff has been shown to be associated with the runoff loss of P in watersheds (Fleming and Cox, 2001). Similarly, since the presence of dissolved organic matter or soluble C often increases trace metal adsorption (Kabata-Pendias, 2001), trace metal mobility can potentially increase if the dissolved organic matter or soluble C becomes mobile; thus, DOC-facilitated metal migration may occur (Dunnivant et al., 1992; Zhou and Wong, 2001). Flow-weighted mean runoff DOC concentrations were highly positively correlated ($r > 0.85$) with both FWM runoff K and TDP concentrations indicating that, as runoff DOC concentrations increased, runoff K and TDP concentrations also increased. McDowell and Sharpley (2003) also showed that the concentration of dissolved reactive P in runoff increased as the C concentration in both dairy manure and poultry litter increased. In contrast, FWM runoff DOC concentrations were highly negatively correlated ($r > -0.9$) with both FWM runoff Ca and Fe concentrations indicating that, as runoff DOC concentrations increased, runoff Ca and Fe concentrations decreased. Runoff DOC losses were also highly positively correlated ($r > 0.85$) with runoff losses of NH_4–N, K, Mg, Na, TDP, Cd, Cr, Cu, Mn, Se, and Zn.

As Vadas et al. (2004) described for dissolved P, when the runoff-to-precipitation ratio is relatively high, as was the case with the low litter treatment compared to the other two litter treatments, it can be expected that runoff concentrations would also be relatively greater, which was also the case for most of the soluble constituents evaluated in this study. The numerically smaller runoff from the high litter treatment and control likely resulted in greater water infiltration, which also likely resulted in greater infiltration of soluble litter constituents and lower runoff concentrations and loads from the high litter treatment and control compared to the low litter treatment. Regardless of the litter rate, only a small fraction of the litter-applied nutrients and metals were accounted for in runoff, which indicates a significant proportion infiltrated the soil to possibly bind with/to soil, be extracted by plants, and/or leach.

This brings about a management dilemma in terms of how to apply broiler litter or other organic soil amendments. Surface application alone could result in increased soluble C runoff concentrations and losses with concomitant increased P and trace metals in runoff, as evidenced by the results of this study. However, though incorporation of litter into the soil may reduce runoff losses of near-surface soluble C and other associated nutrients and/or trace metals, incorporation typically requires severe disturbance of the soil surface, which generally enhances soil erosion, leading to increased surface transport of sediment and sediment-bound constituents, such as P, particularly in regions like the Ozark Highlands where many areas that receive annual litter applications have slopes >5%.

Sloping lands that receive poultry litter additions are also hydrologically connected to nearby surface waters (i.e., rivers and lakes) via potential runoff. Edwards and Daniel (1993) reported that 2 to 7% of the total P land-applied in poultry litter is susceptible to runoff, of which up to 80% can be in the biologically active form to contribute to immediate use and increased algal growth, the precursors to eventual eutrophication of surface waters. However, aside from soluble P and, to some extent, soluble N forms because of their role in eutrophication, the runoff potential of many other plant nutrients and even many trace metals

Table 7. Broiler litter application rate (0 [Control], 5.6 [Low], and 11.2 [High] Mg dry litter ha^{-1}) effects on 4-yr cumulative runoff losses. Treatment means were separated by least significant difference (LSD) at the 0.05 level when appropriate. Coefficients of variation (CV) are also reported.

Runoff property (g ha^{-1})†	Litter treatment			CV (%)	*P*-value	$LSD_{0.05}$
	Control	Low	High			
NO_3–N	14	28	38	48	0.26	–
NH_4–N	74	160	118	35	0.09	–
DOC	666	1415	708	46	0.18	–
Ca	357	413	416	17	0.76	–
K	652	1507	875	48	0.15	–
Mg	68	123	86	33	0.24	–
Na	108	178	134	28	0.31	–
TDP	90	222	129	46	0.11	–
As	1.3	2.5	3.6	53	0.23	–
Cd‡	0.016a	0.034b	0.017a	42	0.01	0.007
Cr	0.17	0.21	0.17	15	0.44	–
Cu	0.40	0.66	0.47	28	0.29	–
Fe‡	10a	6a	17b	44	0.03	5.7
Mn	3.4	5.0	2.8	36	0.30	–
Ni	0.07	0.13	0.12	34	0.21	–
Se	0.35	0.45	0.36	15	0.06	–
Zn	7.0	11	8.0	33	0.46	–

† DOC, dissolved organic carbon; TDP, total dissolved phosphorus.

‡ Means in the same row followed by different letters are significantly different.

from areas with land-applied poultry litter have received relatively little attention but are still key aspects of surface water quality that are important to characterize (Wood et al., 1999).

The USGS monitored surface and groundwater quality between 1992 and 1995 at selected sites in the Ozark Highlands region (Petersen et al., 1998). Table 6 summarizes concentration ranges of selected nutrients in surface and groundwater and of selected trace metals in bed sediments. It is interesting to note that 4-yr FWM NO_3–N runoff concentrations measured in this study (Table 5) were within the range detected in the Ozark Highlands, but that 4-yr FWM soluble P runoff concentrations (Table 5) were three to five times greater than that observed in the regional survey (Table 6). Furthermore, though trace metal water concentrations were not reported in the USGS water quality survey, trace metals were detected at somewhat high concentrations at times in the bed sediments of surface waters (Petersen et al., 1998), likely indicating an external source of trace metals to these waterways. It was also concluded that surface and groundwater nutrient concentrations were greater when the surrounding land use was agricultural than when it was forest (Petersen et al., 1998).

Based on a water quality survey of Ozark Highland streams conducted by Ekka et al. (2006), annual and 4-yr mean soluble P runoff concentrations observed in this study were 10 to 50 times greater than background P concentrations upstream of wastewater treatment plants (0.02 to 0.12 mg P L^{-1}) but were more similar to P concentrations measured downstream of the same wastewater treatment plants. These results indicate that the nonpoint-source runoff P concentrations from litter-amended pasturelands can be of the same order of magnitude as highly regulated point-source effluent discharges.

Of the runoff studies that have been conducted in the Ozark Highlands to investigate the magnitude of potential nonpoint-source nutrient enrichment of surface waters, most have been rainfall simulation studies conducted on the same Captina silt-loam soil as this study (Edwards and Daniel, 1993, 1994; Shreve et al., 1995; Moore et al., 1998; Sauer et al., 1999). Runoff concentrations immediately after, and at times several weeks to months after, litter application and application of simulated rainfall are generally greater than the concentrations reported in this study in response to natural precipitation (Table 5; Edwards and Daniel, 1993, 1994; Shreve et al., 1995; Moore et al., 1998; Sauer et al., 1999; Pote et al., 2003). Thus, despite at times resulting in similar conclusions and interpretations, clearly there is a discrepancy between the magnitude of runoff concentrations and losses from rainfall simulation studies and those from natural precipitation patterns. It could be argued that long-term results from natural precipitation events better characterize ecosystem response and the potential environmental significance of runoff from broiler litter-amended soil.

Summary and Conclusions

Broiler litter application, or the lack thereof, had varying effects on FWM concentrations, and runoff quantity did not always dictate greater or lower FWM concentrations of nutrients or trace metals observed in runoff. Though dilution may have occurred, numerically greater runoff from particular rainfall events could have actually increased nutrient or metal loads in runoff. Furthermore, periods of rapid vegetative growth and organic matter contributions via broiler litter applications likely influenced litter treatment differences more than originally thought.

Environmentally significant results were observed for FWM concentrations of TDP and As. All annual FWM TDP concentrations from all treatments exceeded the minimum P concentrations required for algae growth. In addition, annual FWM As concentrations exceeded the As MCL for drinking water 75% of the time for all treatments, which in 3 of 4 yr were greater from the low and/or high litter treatments than from the unamended control. Flow-weighted mean concentrations of all other metals evaluated were below the MCL for drinking water, and, currently, there is no MCL specified for the other plant macronutrients reported in this study.

The tendency for increased annual runoff losses of some soluble trace metals, particularly As, Fe, and Se, after repeated broiler litter applications is somewhat alarming. The exposure to the environment of increasing metal concentrations, and their subsequent mobility either relatively quickly due to runoff or somewhat more slowly, but eventually, due to leaching, is an important negative environmental consequence of the land application of broiler litter that cannot be ignored and requires further monitoring.

Results from this 4-yr study indicate that reducing broiler litter application may reduce runoff losses of some potentially environmentally harmful plant macronutrients and trace metals (i.e., P, As, and Fe). However, eliminating broiler litter application completely, as represented by the control treatment in this study, may still lead to years of nutrient and metal enriched runoff (i.e., TDP, Cu, Cr, Fe, Mn, Ni, and Zn) due to the soil's ability to concentrate, retain, and recycle nutrients and trace metals near the soil surface.

References

Allen, M.B., and D.I. Arnon. 1955. Studies on nitrogen fixing blue-green algae: I. Growth and nitrogen fixation by *Anabaena cylindrica* Lemm. Plant Physiol. 30:366–372.

Brye, K.R., and A.L. Pirani. 2006. Metal uptake by tall fescue (*Festuca arundinacea*) as affected by poultry litter application. Grass Forage Sci. 61:192–199.

Brye, K.R., J.M. Norman, L.G. Bundy, and S.T. Gower. 1999. An equilibrium tension lysimeter for measuring drainage through soil. Soil Sci. Soc. Am. J. 63:536–543.

Chapman, S.L. 1996. Soil and solid poultry waste nutrient management and water quality. Poult. Sci. 75:862–866.

Chu, S.P. 1943. The influence of mineral composition of the medium on the growth of planktonic algae: II. The influence of the concentration of inorganic nitrogen and phosphate phosphorus. J. Ecol. 32:109–148.

DeLaune, P.B., P.A. Moore, D.K. Carman, A.N. Sharpley, B.E. Haggard, and T.C. Daniel. 2004a. Development of a phosphorus index for pastures fertilized with poultry litter- factors affecting phosphorus runoff. J. Environ. Qual. 33:2183–2191.

DeLaune, P.B., P.A. Moore, D.K. Carman, A.N. Sharpley, B.E. Haggard, and T.C. Daniel. 2004b. Evaluation of the phosphorus source component in the Phosphorus Index for pastures. J. Environ. Qual. 33:2192–2200.

Dunnivant, F.M., P.M. Jardine, D.L. Taylor, and J.F. McCarthy. 1992. Co-transport of cadmium and hexachlorobiphenyl by dissolved organic carbon through columns containing aquifer material. Environ. Sci. Technol. 26:360–368.

Edwards, D.R., and T.C. Daniel. 1992. Environmental impacts of on-farm poultry waste disposal- A review. Bioresour. Technol. 41:9–33.

Edwards, D.R., and T.C. Daniel. 1993. Effects of poultry litter application rates and rainfall intensity on quality of runoff from fescuegrass plots. J. Environ. Qual. 22:361–365.

Edwards, D.R., and T.C. Daniel. 1994. Quality of runoff from fescuegrass plots treated with poultry litter and inorganic fertilizer. J. Environ. Qual. 23:579–584.

Ekka, S.A., B.E. Haggard, M.D. Matlock, and I. Chaubey. 2006. Dissolved phosphorus concentration and sediment interactions in effluent-dominated Ozark streams. Ecol. Eng. 26:375–391.

Finlay-Moore, O., P.G. Hartel, and M.L. Cabrera. 2000. 17β-estradiol and testosterone in soil and runoff from grasslands amended with broiler litter. J. Environ. Qual. 29:1604–1611.

Fleming, N.K., and J.W. Cox. 2001. Carbon and phosphorus losses consideration of scale effects using the conceptual from dairy pasture in South Australia. Aust. J. Soil Res. 39:969–978.

Franklin, D.H., M.L. Cabrera, and V.H. Calvert. 2006. Fertilizer source and soil aeration effects on runoff volume and quality. Soil Sci. Soc. Am. J. 70:84–89.

Franzluebbers, A.J. 2002. Water infiltration and soil structure related to organic matter and its stratification with depth. Soil Tillage Res. 66:197–205.

Gupta, G., J. Borowiec, and J. Okoh. 1997. Toxicity identification of poultry litter aqueous leachate. Poult. Sci. 76:1364–1367.

Gupta, G., and S. Charles. 1999. Trace elements in soils fertilized with poultry litter. Poult. Sci. 78:1695–1698.

Haggard, B.E., P.A. Moore, I. Chaubey, and E.H. Stanley. 2003. Nitrogen and phosphorus concentrations and export from an Ozark Plateau catchment in the United States. Biosyst. Eng. 86:75–85.

Harper, D., W.W. Phillips, and G.J. Haley. 1969. Soil Survey of Washington County, Arkansas. U.S. Gov. Print. Office, Washington, DC.

Hershfield, D.M. 1961. Rainfall frequency atlas of the United States. Tech. Pap. No. 40. Weather Bureau, U.S. Dep. of Commerce, Washington, DC.

Hileman, L.H. 1973. Response of orchardgrass to broiler litter and commercial fertilizer. Rep Ser. 207, Ark. Agric. Expt. Stn., Univ. Ark., Fayetteville, AR.

Huneycutt, H.G., C.P. West, and J.M. Phillips. 1988. Responses of bermudagrass, tall fescue, and tall fescue-clover to broiler litter and commercial fertilizer. Bull. 913, Ark. Agric. Expt. Stn., Univ. Ark., Fayetteville, AR.

Jones, F.T. 2007. A broad view of arsenic. Poult. Sci. 86:2–14.

Kabata-Pendias, A. 2001. Trace elements in soils and plants. Third ed. CRC Press, Boca Raton, FL.

Kingery, W.L., C.W. Wood, D.P. Delaney, J.C. Williams, and G.L. Mullins. 1994. Impact of long-term land application of broiler litter on environmentally related soil properties. J. Environ. Qual. 23:139–147.

Kingery, W.L., C.W. Wood, D.P. Delaney, J.C. Williams, G.L. Mullins, and E. Van-Santen. 1993. Implications of long-term land applications of poultry litter on tall fescue pastures. J. Prod. Agric. 6:390–395.

Kunkle, W.E., L.E. Carr, T.A. Carter, and E.H. Bossard. 1981. Effect of floor type on the levels of nutrients and heavy metals in poultry litter. Poult. Sci. 60:1160–1164.

Likens, G.E., F.H. Bormann, R.S. Pierce, J.S. Eaton, and N.M. Johnson. 1977. Biogeochemistry of a forested ecosystem. Springer-Verlag, New York.

Loague, K., and R.H. Abrams. 2001. Stochastic-conceptual analysis of near-surface hydrological response. Hydrol. Processes 15:2715–2728.

Menjoulet, B. 2007. Metal and plant nutrient runoff from poultry litter-amended tall fescue. M.S. thesis. Univ. of Arkansas, Fayetteville.

Matera, V., and I. Le Hécho. 2001. Arsenic behavior in contaminated soils: Mobility and speciation. p. 207. *In* H.M. Selim and D.L. Sparks (ed.) Heavy metals release in soils. CRC Press, Boca Raton, FL.

McDowell, R.W., and A.N. Sharpley. 2003. The effects of soil carbon on phosphorus and sediment loss from soil trays by overland flow. J. Environ. Qual. 32:207–214.

Mitchell, C.C., and S. Tu. 2006. Nutrient accumulation and movement from poultry litter. Soil Sci. Soc. Am. J. 70:2146–2153.

Moore, P.A., T.C. Daniel, J.T. Gilmour, B.R. Shreve, D.R. Edwards, and B.H. Wood. 1998. Decreasing metal runoff from poultry litter with aluminum sulfate. J. Environ. Qual. 27:92–99.

NOAA. 2006. Washington County, Arkansas Climatology. National Oceanic and Atmospheric Administration, Washington, DC. Available at http://www.srh.noaa.gov/tsa/climate/washington.html (verified 26 Sept. 2008).

Peters, J. (ed.). 2003. Recommended methods of manure analysis. Univ. of Wisconsin–Extension Publ. A3769, Madison, WI.

Petersen, J.C., J.C. Adamski, R.W. Bell, J.V. Davis, S.R. Femmer, D.A. Freiwald, and R.L. Joseph. 1998. Water quality in the Ozark Plateaus, Arkansas, Kansas, Missouri, and Oklahoma, 1992–95. U.S. Geol. Surv. Circ. 1158. USGS, Washington, DC. Available at http://pubs.usgs.gov/circ/circ1158/index.html (verified 15 Jan. 2009).

Pierson, S.T., M.L. Cabrera, G.K. Evanylo, H.A. Kuykendall, C.S. Hoveland, M.A. McCann, and L.T. West. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. J. Environ. Qual. 30:1784–1789.

Pirani, A.L. 2005. Leaching of plant nutrients and heavy metals from poultry litter-amended tall fescue pasture. M.S. thesis. Univ. of Arkansas, Fayetteville, AR.

Pirani, A.L., K.R. Brye, T.C. Daniel, B.E. Haggard, E.E. Gbur, and J.D. Mattice. 2006. Soluble metal leaching from a poultry litter-amended Udult under pasture vegetation. Vadose Zone J. 5:1017–1034.

Pirani, A.L., K.R. Brye, B.E. Haggard, T.C. Daniel, and J.D. Mattice. 2007. Broiler litter rate effects on nutrient leaching from soil under pasture vegetation in the Ozark Highlands. Soil Sci. 172:1001–1018.

Pote, D.H., W.L. Kingery, G.E. Aiken, F.X. Han, P.A. Moore, and K. Buddington. 2003. Water-quality effects of incorporating poultry litter into perennial grassland soils. J. Environ. Qual. 32:2392–2398.

Rhoton, F.E., M.J. Shipitalo, and D.L. Lindbo. 2002. Runoff and soil loss from midwestern and southeastern US silt loam soils as affected by tillage practice and soil organic matter content. Soil Tillage Res. 66:1–11.

Sauer, T.J., T.C. Daniel, P.A. Moore, K.P. Coffey, D.R. Nichols, and C.P. West. 1999. Poultry Litter and grazing waste effects on runoff water quality. J. Environ. Qual. 28:860–865.

Sawyer, C.N. 1947. Fertilization of lakes by agricultural and urban drainage. N. Engl. Water Works Assoc. J. 61:109–127.

Saxton, K.E., W.J. Rawls, J.S. Romberger, and R.I. Papendick. 1986. Estimating generalized soil-water characteristics from texture. Soil Sci. Soc. Am. J. 50:1031–1036.

Schroeder, P.D., D.E. Radcliffe, and M.L. Cabrera. 2004. Rainfall timing and poultry litter application rate effects on phosphorus loss in surface runoff. J. Environ. Qual. 33:2201–2209.

Sharpley, A.N., S.J. Smith, and R. Bain. 1993. Effect of poultry litter application on the nitrogen and phosphorus content of Oklahoma soils. Soil Sci. Soc. Am. J. 57:1131–1137.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30–35.

Shreve, B.R., P.A. Moore, T.C. Daniel, D.R. Edwards, and D.M. Miller. 1995. Reduction of phosphorus in runoff from field-applied poultry litter using chemical amendments. J. Environ. Qual. 24:106–111.

Sims, J.T., and D.C. Wolf. 1994. Poultry waste management: Agricultural and environmental issues. Adv. Agron. 52:1–83.

Sistani, K.R., D.A. Mays, and R.A. Dawkins. 2006. Tall fescue fertilized with alum-treated and untreated broiler litter: Runoff, soil, and plant nutrient content. J. Sustain. Agric. 28:109–119.

Slaton, N.A., K.R. Brye, M.B. Daniels, T.C. Daniel, R.J. Norman, and D.M. Miller. 2004. Nutrient input and removal trends for agricultural soils in nine geographic regions in Arkansas. J. Environ. Qual. 33:4606–4615.

Stephenson, A.H., T.A. McCaskey, and B.G. Ruffin. 1990. A survey of broiler litter composition and potential value as a nutrient resource. Biol. Wastes 34:1–9.

Tarkalson, D.D., and R.L. Mikkelsen. 2004. Runoff phosphorus losses as related to phosphorus source, application method, and application rate on a Piedmont soil. J. Environ. Qual. 33:1424–1430.

Tarkalson, D.D., and R.L. Mikkelsen. 2007. Phosphorus losses in runoff after application of litter from broilers fed high-available phosphorus corn feed. Commun. Soil Sci. Plant Anal. 38:1805–1814.

Tomlinson, P.J., M.C. Savin, and P.A. Moore. 2007. Phosphatase activities in soil after repeated untreated and alum-treated poultry litter applications. Biol. Fertil. Soils (in press).

Tucker, M.R. 1992. Determination of phosphorus by Mehlich-3 extraction. p. 6–8. *In* S.J. Donohue (ed.) Soil and media diagnostic procedures for the Southern Region of the United States. Virginia Agricultural Experiment Station. Series Bull. 374. Blacksburg, VA.

University of Arkansas Cooperative Extension Service (UA-CES). 2002. Improving poultry litter management and carcass disposal. Misc. Publ.-Fact Sht. 8. UA-CES, Little Rock, AR.

University of Arkansas Cooperative Extension Service (UA-CES). 2006. Improving poultry litter management and carcass disposal. Available at http://www.arnatural.org/environmental_management/eqip4/fact8.htm (verified 26 Sept. 2008). UA-CES, Little Rock, AR.

USDA-NASS. 2002. 2002 Census of agriculture: State and county profiles. Available at http://www.agcensus.usda.gov/Publications/2002/County_Profiles/Arkansas/index.asp (verified 15 Jan. 2009). USDA-NASS, Washington, DC.

USDA-NASS. 2007. Broilers: Production and value of production by year, US. Available at http://www.nass.usda.gov/Charts_and_Maps/Poultry/brprvl.asp (verified 26 Sept. 2008). USDA-NASS, Washington, DC.

USDA-NRCS. 2004. Official soil series descriptions. Available at http://ortho.ftw.nrcs.usda.gov/cgi-bin/osd/osdname.cgi (verified 26 Sept. 2008). USDA-NRCS, Washington, DC.

USDA-NRCS. 2007. Web soil survey. Available at http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx (verified 26 Sept. 2008). USDA-NRCS, Washington, DC.

USEPA. 1996. Test methods for evaluating solid wastes, physical/chemical methods, 3rd ed. Available at http://www.epa.gov/epaoswer/hazwaste/test/pdfs/6010b.pdf (verified 26 Sept. 2008). USEPA, Washington, DC.

USEPA. 2001a. Ambient water quality criteria recommendations: Lakes and reservoirs in nutrient ecoregion 5. Report EPA 822-B-01-010. USEPA, Washington, DC.

USEPA. 2001b. Ambient water quality criteria recommendations: Rivers and streams in nutrient ecoregion 5. Report EPA 822-B-01-014. USEPA, Washington, DC.

USEPA. 2006. Drinking water contaminants. Available at http://www.epa.gov/safewater/contaminants/index.html#mcls (verified 26 Sept. 2008). USEPA, Washington, DC.

USGS. 1999. The quality of our nation's waters-Nutrients and pesticides. Circ. 1225. USGS, Reston, VA. 82 p.

Vadas, P.A., P.J.A. Kleinman, and A.N. Sharpley. 2004. A simple method to predict dissolved phosphorus in runoff from surface-applied manures. J. Environ. Qual. 33:749–756.

Vervoort, R.W., D.E. Radcliffe, M.L. Cabrera, and M. Latimore. 1998. Field-scale nitrogen and phosphorus losses from hayfields receiving fresh and composted broiler litter. J. Environ. Qual. 27:1246–1254.

Walker, J.B. 1953. Inorganic micronutrient requirements of *Chlorella*. I. Requirements for calcium (or strontium), copper and molybdenum. Arch. Biochem. Biophys. 46:1–11.

Wood, B.H., C.W. Wood, K.H. Yoo, K.S. Yoon, and D.P. Delaney. 1999. Seasonal surface runoff losses of nutrients and metals from soils fertilized with broiler litter and commercial fertilizer. J. Environ. Qual. 28:1210–1218.

Zhou, L.X., and J.W.C. Wong. 2001. Effect of dissolved organic matter from sludge and sludge compost on soil copper sorption. J. Environ. Qual. 30:878–883.

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Delineating runoff processes and critical runoff source areas in a pasture hillslope of the Ozark Highlands

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Abstract:

The identification of runoff contributing areas would provide the ideal focal points for water quality monitoring and Best Management Practice (BMP) implementation. The objective of this study was to use a field-scale approach to delineate critical runoff source areas and to determine the runoff mechanisms in a pasture hillslope of the Ozark Highlands in the USA. Three adjacent hillslope plots located at the Savoy Experimental Watershed, north-west Arkansas, were bermed to isolate runoff. Each plot was equipped with paired subsurface saturation and surface runoff sensors, shallow groundwater wells, H-flumes and rain gauges to quantify runoff mechanisms and rainfall characteristics at continuous 5-minute intervals. The spatial extent of runoff source areas was determined by incorporating sensor data into a geographic information-based system and performing geostatistical computations (inverse distance weighting method). Results indicate that both infiltration excess runoff and saturation excess runoff mechanisms occur to varying extents (0–58% for infiltration excess and 0–26% for saturation excess) across the plots. Rainfall events that occurred 1–5 January 2005 are used to illustrate the spatial and temporal dynamics of the critical runoff source areas. The methodology presented can serve as a framework upon which critical runoff source areas can be identified and managed for water quality protection in other watersheds.

KEY WORDS runoff; saturation excess runoff; infiltration excess runoff; runoff source area; GIS

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INTRODUCTION

Storm runoff generation is a non-linear process that has surface and subsurface components. This has led to a number of runoff generation concepts, namely the Horton (1933) overland flow concept, the partial area concept (Betson, 1964) and the variable source area concept (Hewlett and Hibbert, 1967). While the runoff generation process at a particular location within a watershed could be a combination of these processes depending on climate, geology, topography, soil characteristics and rainfall patterns, studies have found that limited (less than 10%) portions of a watershed can contribute disproportionately large runoff amounts at the downslope end of the watershed (Freeze, 1974).

Large amounts of storm runoff from a catchment has implications for water quality, flood control and watershed management. For example, storm runoff plays a major role in phosphorous (P) transport and diffuse P pollution is a major contributor to freshwater systems. The role of P in accelerating eutrophication in freshwater systems was recognized over three decades ago (Schindler *et al.*, 1971), and as a result, P transport has become a focus of water quality research. For hillslope watersheds, Pionke *et al.* (1997) reported that approximately 10% of a watershed area may contribute up to 90% of the annual P loads from that watershed. Although the relationship between runoff generation and P transport has been recognized for a decade (Zollweg *et al.*, 1995; Gburek and Sharpley, 1998), there is a need to accurately identify and delineate critical areas (known as runoff source areas) of a watershed that may contribute heavily to runoff and nutrient transport. Identification of critical runoff source areas has many levels of importance to water resource managers and is of critical importance to the south-eastern USA. Runoff source areas can serve as focus areas for water quality monitoring, watershed management, and Best Management Practice (BMP) implementation (Walter *et al.*, 2000).

The idea of delineating runoff source areas in a watershed is not new; Dunne *et al.* (1975) suggested repeated field mapping methods for catchment areas less than 8 km^2. Dunne *et al.* (1975) presented methodologies that included the use of topographic, soil and vegetative indicators to determine the extent of the runoff-producing zones for a north-eastern Vermont catchment. However, the Dunne *et al.* (1975) methodology lacked a continuous time scale, a rather important factor in the determination of hydrological parameters that influence storm runoff (Srinivasan *et al.*, 2002). Anderson and Burt

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(1978) presented instrumentation methods for determining runoff source areas by measuring field soil water potential. Holden and Burt (2003) mapped runoff depth for blanket peat catchments of the northern Pennines of the UK. Zollweg (1996) developed sensors to determine the soil saturation level within a 26 ha area watershed in the north-eastern USA. Srinivasan *et al.* (2000) improved upon the Zollweg sensors by automation and coupled them with surface runoff sensors to measure the temporal dynamics of runoff source areas in east-central Pennsylvania.

In this paper, a field-scale methodology is presented that identifies critical runoff source areas and dominant runoff mechanisms in a pasture watershed. This project was accomplished through the measurement of surface and subsurface field and watershed characteristics. The overall goal of this project was to understand the complexity of runoff production from a pasture watershed that overlies mantled karst geologic features.

SITE DESCRIPTION

This study was conducted on three adjacent 23 m × 23 m hillslope-plots established in Basin 1 (Figure 1) of the 1250-ha Savoy Experimental Watershed (SEW) located in north-west Arkansas, USA. The SEW serves as a multidisciplinary research site which is being used for long-term hydrologic monitoring (Brahana *et al.*, 2005). Basin 1 is a 147 ha sub-basin that drains into the Illinois River. The Illinois River is a transboundary river that originates approximately 24 km south-west of the city of Fayetteville, Arkansas and flows in a north-westerly direction into Oklahoma. Land use within the Illinois River watershed is approximately 58% pasture, 36% forest and 6% urban (Soerens *et al.*, 2003). The fact that Arkansas and north-west Arkansas in particular is a leader in poultry production in the USA, has led to practices such as the application of poultry litter to the soil for a long time. Over the past two decades there has been considerable controversy between the states of Oklahoma and Arkansas over the source of elevated P loadings in the Illinois River. Thus, there is a need to develop methodologies for identifying runoff- producing areas in this basin. Long-term (30 year) average annual precipitation measured in the city of Fayetteville (approximately 12 km east of the study location) was 117 cm. A low average monthly precipitation of 5 cm occurs in January and an average monthly high of 13 cm occurs in June. Winters are relatively short, with brief periods of snow cover and an average January temperature of 1 °C, whereas summers are warm and humid with an average July temperature of 26 °C (NOAA, 2002).

Land use of the SEW is representative of a typical pasture-dominated agricultural field in the Ozark Highlands. There are six major soils present immediately near and within Basin 1 with the Clarksville cherty silt loam (12–60% slope, loamy-skeletal, siliceous, semiactive, mesic Typic Paleudults) and Nixa cherty silt loam (3–8% slope, loamy-skeletal, siliceous, active, mesic Glossic Fragiudults) accounting for 79% of the area (Sauer and Logsdon, 2002). Other soils include the Pickwick silt loam (3–8% slope, fine-silty, mixed, semiactive, thermic Typic Paleudults), Razort silt loam (fine-loamy, mixed, active, mesic Mollic Hapludalfs) and Razort gravelly silt loam. The two principal soil series within the

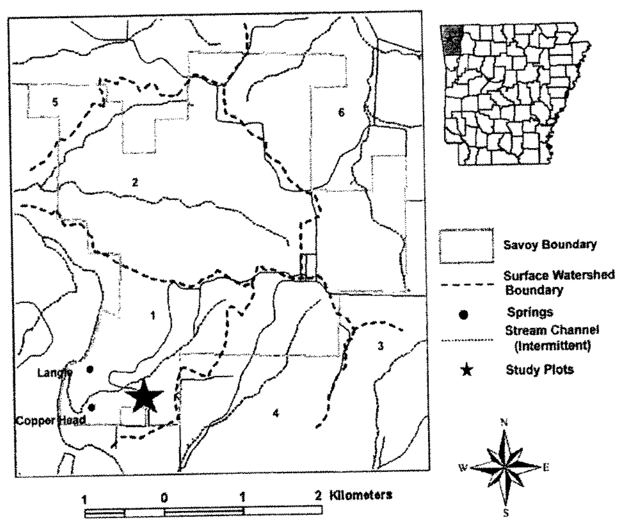

Figure 1. Location of the study site at the Savoy Experimental Watershed (SEW) in north-west Arkansas and study plots. Numbers represent surface water drainage basins

study plots are Captina silt loam (3–6% slope, fine-silty, siliceous, active, mesic Typic Fragiudults) and Nixa cherty silt loam. The Captina silt loam soil is typically formed on stream terraces and has strong brown subsoil that is 25–50 cm thick. Soils of the Nixa cherty silt loam typically have slow permeable fragipans that occur at the 36–60 cm depth (Harper *et al.*, 1969).

INSTRUMENTATION

The three 23 m × 23 m plots (labelled plot 1, plot 2, and plot 3 in Figure 2) were bermed to isolate runoff and to prevent run-on from the upslope area. A perimeter fence with gates was constructed around the field boundary since the surrounding pastures are typically grazed. A grid of paired subsurface saturation sensors (SSS) and surface runoff sensors (SRS) (henceforth called saturation and runoff sensor, respectively) were installed at 33 points on the plots (12 on plot 1, 11 on plot 2 and 10 on plot 3 (Figure 2)). In this project, the sensors from Srinivasan *et al.* (2000) were used to quantify the dominant runoff mechanisms and identify runoff contributing areas. The saturation sensors are printed circuit boards with sensing pins that indicate the level of soil saturation at preset depths (1 cm, 5 cm, 10 cm, 20 cm, 31 cm and 46 cm). The runoff sensors are miniature v-notch weirs made of 2 mm thick galvanized sheet metal with a sensor pin and ground pin set 2 cm apart and 2·5 cm away from the v-notch, at the same level as the bottom of the v-notch. The runoff sensor

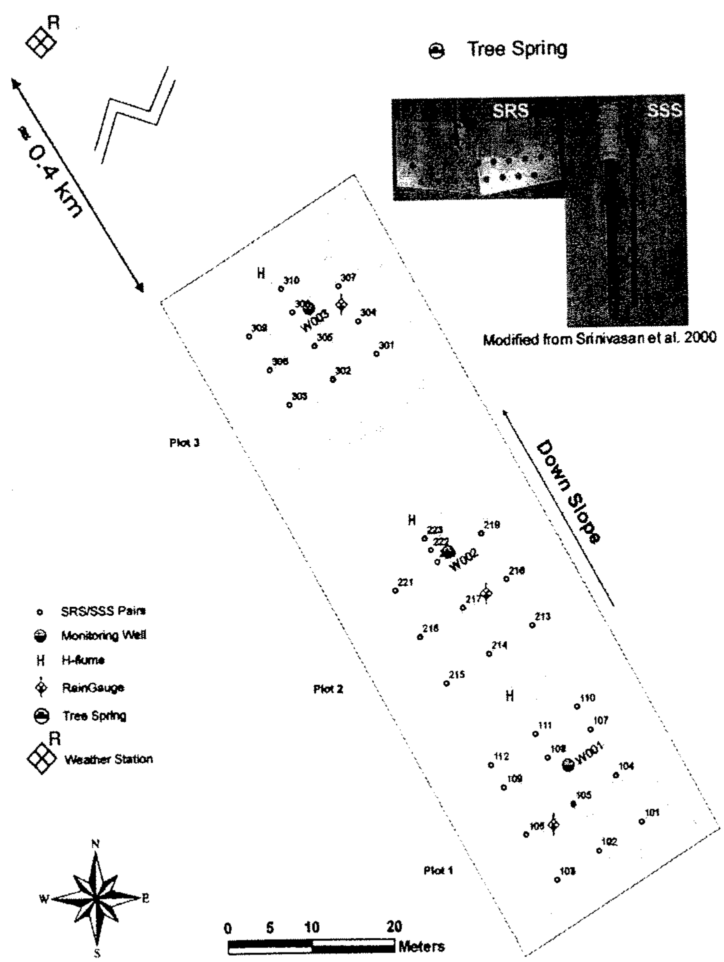

Figure 2. Site layout showing orientation and layout of data collection infrastructure including weather station, spring, rain gauges, H-flumes, surface runoff sensors (SRS) and subsurface sensors (SSS) and monitoring wells on plots 1 through 3

operates on a 'yes-no' basis to indicate the presence or absence of surface runoff. Full descriptions of the sensors are presented by Srinivasan *et al.* (2000).

These plots were also instrumented with a 0·305 m H-flume at the downslope end to measure the total volume of runoff from each plot. A Keller pressure transducer (Model 173, Keller America Inc., Newport News, Virgina) was used to record water depth at each flume, and the pre-determined stage–discharge relation for the flume was used to estimate runoff. It must be noted here that there is no plot-to-plot surface interaction of runoff. A 6 cm polyvinyl chloride (PVC) pipe connected a 114 L runoff collector to the H-flume at the downslope end of each plot to route surface runoff and prevent concentrated flow getting into the down slope plots. Runoff samples were collected for water quality analysis and the runoff collector subsequently emptied after each rainfall event.

A tipping bucket rain gage (HOBO RG2 model) was installed at each plot to record the spatial extent of rainfall variability. A shallow groundwater well (with a Keller pressure transducer) was installed (0·6–0·7 m deep) near each saturation sensor upslope of each H-flume to monitor depth to ground water. Locations of all infrastructures are shown in Figure 2.

All instruments were connected to a series of multiplexers and logged at 5 min intervals with a Campbell CR-23X data logger on plots 1 and 2, and a Campbell CR-10X (Campbell Scientific Inc., Logan Utah) on plot 3. Rainfall data on each plot were compared with that of a nearby weather station located approximately 0·4 km north of the study site. A spring in the northeastern corner of the plots was instrumented with a 0·152 m H-flume and a data logger.

METHODS

Topographic and Geophysical Surveys

A topography survey of the field area was determined using a high precision Global Positioning System (GPS) to obtain elevation data. The GPS survey was performed from 18 February 2004 through 23 March 2004 by establishing 1 m grid points across the plots. The grid points were georeferenced with a survey grade GPS (Leica 500 SkiPro) to the WGS 1984 coordinate system. The data obtained were processed in GIS software to obtain a 1 m resolution digital elevation model (DEM) of the field.

Surface geophysical investigations of the soil and bedrock were performed in March 2004 to characterize the subsurface attributes beneath the plots (Ernenwein and Kvamme, 2004). Two techniques (ground penetrating radar (GPR) and electrical resistance) were used to determine the underlying subsurface characteristics. In addition to seismic methods, GPR and resistance methods are the most intensively utilized techniques to map bedrock depth and integrity (Ernenwein and Kvamme, 2004).

Ground penetrating radar data collection was accomplished with a Geophysical Survey Systems International (GSSI) Subsurface Imaging Radar (SIR) 2000 system and a 400 MHz antenna. The data collection consisted of dragging the antenna across the ground surface north to south. Reflections were recorded at a rate of 32 scans per second through a range of 80 ns (two-way travel time). The GPR measurements resulted in 23 transect profiles (also known as radargrams) across the plots at 12 different depths across the hillslope. Each transect measurement was taken approximately 1 m apart and the depth measurements were at approximately 0·9 m intervals.

The resistivity survey involved introducing a weak electrical current into the ground and measuring the potential voltage. The amount of electric current that flows through the soil is directly related to soil moisture, clay content, and solutes present in the soil and rock. Electrical resistance measurements were taken every 0·5 m along parallel lines separated by 1 m with a Geoscan Research RM-15 meter. A twin-probe array with a mobile probe separation of 1 m was used to target a depth of about 1 m below the surface of the plots.

Infiltration measurements and soil physical properties

Ponded infiltration measurements were performed under relatively dry antecedent soil moisture conditions (volumetric water content $<0·34$ cm^3 cm^{-3}). Measurements were taken on 14 June 2005 and 15 June 2005 with a double ring infiltrometer (0·15 m inner ring diameter and 0·2-m height). Three random infiltrometer measurements were made and saturated hydraulic conductivity was estimated from the final infiltration rates for each plot. The locations of the infiltration measurements were not documented. Soil core samples were taken from the 0–5 cm depth at 65 points across the plots (see Figure 6 for sampling location) to determine soil bulk density and porosity. Measurements at the 65 points were imported into a GIS software and interpolated (using an inverse weighted distance method) to derive the spatial distribution of soil properties across the plots. Statistical analyses were performed on measured values and infiltration rates using analysis of variance (ANOVA). Mean infiltration rates and bulk density were separated using Fisher's protected least significant difference test (LSD) at $P = 0·05$.

Runoff mechanisms on the plots

Each runoff event was analysed to quantify the occurrence of saturation excess versus infiltration excess runoff on each plot. Saturation excess runoff was operationally defined as cases where the runoff sensors indicated runoff while the saturation sensors indicated a saturated soil level within 1 cm of the soil surface (surface saturation). For each event, a runoff response ratio was (RRR) defined as

$$RRR_i = \frac{\sum R_p}{n} \tag{1}$$

where R_p was the number of 5 min occurrences of a particular runoff process at a sensor location, and n was total number of 5 min readings taken (non-runoff events inclusive). The average *RRR* for the period of the study (2004–2005) was used to characterize the dominant runoff mechanisms that occurred.

Runoff-contributing areas

Runoff-contributing areas were quantified by interpolating runoff location points as indicated by the runoff sensors, either 1 or 0. Thus, sensor locations where runoff was known to occur were given a value of 1 and sensor locations that did not indicate runoff were assigned a value of 0. A binary map was then created by applying a threshold of 60% and above to delineate the runoff source areas. It is acknowledged that the 60% threshold is arbitrary; however, this value was arrived at empirically as a result of field observation of the area after storm events. The total percentage area contributing to runoff (A_r) during each event period was calculated as:

$$A_r = \left(\frac{\sum P_R}{P_T}\right) \times 100\% \qquad (2)$$

where P_R is the number of pixels within the delineated runoff contributing area, and P_T is the total number of pixels representing all three plots.

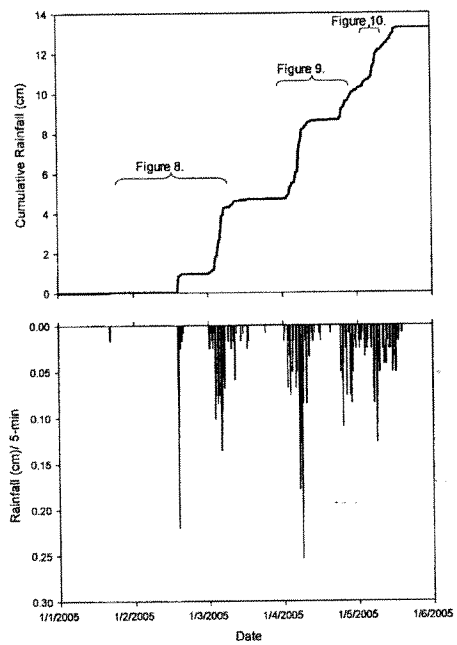

Figure 3. Cumulative rainfall and rainfall intensity in 5 min increments from 1–5 January 2005

DATA ANALYSIS

Rainfall characteristics and the plot runoff response were closely monitored from April 2004 to December 2005. In 2004, 106 precipitation events occurred, totaling 106 cm rainfall. In 2005, 111 rainfall events accounted for 95 cm of rainfall. Precipitation events within the study were defined as any 24 h period with measurable rainfall. Rainfall events that occurred 1–5 January 2005 are presented in detail in this manuscript (Figure 3). These events were selected because they revealed the unique watershed response to rainfall from a period of relatively dry watershed conditions (1–3 January 2005) to relatively wet watershed conditions (4/5 January 2005). Two weeks before the selected events, no precipitation events had occurred, with the exception of two dusting snow events that occurred on 21 and 22 December 2004. These snow events lasted only an hour and were quickly evaporated thereafter.

Sensor data were analysed to determine runoff mechanisms which occur in the field. A paired sensor analysis was used to identify the runoff mechanism that occurred. For example, the surface runoff mechanism was classified as saturation excess runoff if the saturation sensors indicated saturated soil conditions (surface saturation) and the runoff sensors indicated the presence of surface runoff. On the other hand, the runoff mechanism was classified as infiltration excess runoff if the saturation sensors did not indicate saturated conditions near the soil surface while the runoff sensors indicated the presence of surface runoff. A GIS was used to derive the spatial extent of the data and to quantify the spatial distribution of the runoff source areas. A sensor location was considered as contributing to runoff if any of the runoff mechanisms (infiltration excess or saturation excess) occurred at that particular sensor location.

RESULTS AND DISCUSSION

Topographic and geophysical surveys

The resistivity survey (Ernenwein and Kvamme, 2004) revealed a high electrical resistance (>120 Ω) area located in the southern (plot 1) to mid (plot 2) portion of the field (Figure 4). A sharp boundary of the resistance area can be seen across the end of plot 2, running from the north-east to south-west and is thought to represent regolith overlying shallow bedrock (limestone with multiple layers of bedded chert (Figure 5)).

Figure 5A shows a composite map of the GPR data at each depth sampled, and the location of sampled transect 15 (line 15). The details of transect 15 are shown in Figure 5B. In profile form, the GPR data provides a lot of detail of the nature of the subsurface characteristics of the plots. The data confirmed the findings from the resistivity survey, the upper portion of the field (plots 1 and 2) contains abundant contrasting materials (limestone,

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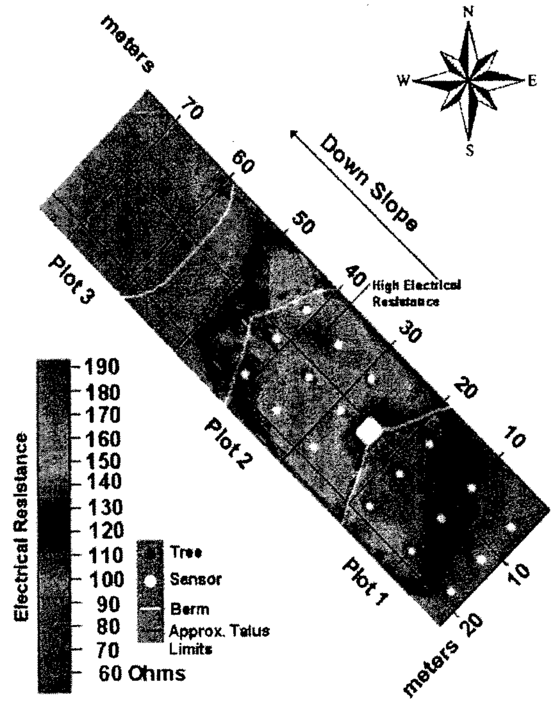

Figure 4. Electrical resistivity survey of study site showing areas of high resistance and selected sensor locations

chert, weathered zones with clay), whereas the lower portion (plot 3) is more homogenous in nature, indicating a weathered and transported regolith with a deeper zone to bedrock (Figure 5B). The resistivity data and GPR data are complementary. Combining both data sets revealed the subsurface structure of the field in great detail and was reinforced by drilling and trenching. Overall, the subsurface structure of plots 1 and 2 are interpreted as containing near-horizontal limestone bedrock with multiple layers of low-permeability chert and clay infilling bedding planes in the limestone; near-vertical fractures and probable, faults transect the plots near the berm separating plots 1 and 2, and near the northern boundary of plot 2 (Figure 5). The chert layers have been truncated by near-surface erosional processes, and are slightly tilted toward the north and west, providing a downward-stepping upper confining unit to plots 1 and 2. The chert layers are interpreted as being downfaulted beneath plot 3 (Figure 5). At Savoy, as in numerous karst areas elsewhere characterized by heterogeneous and anisotropic pathways that underdrain the soil, knowledge of the location of subsurface fractures, joints, bedding planes and distinctive soil horizons is essential to understanding the three-dimensional details of flow pathways that control runoff processes. At the study site, the limestone and chert represents the lowermost part (approximately 5 m) of the Boone Formation; the Boone is stratigraphically underlain by the St. Joe Formation, a pure limestone that is highly susceptible to dissolution and subsurface capture of infiltrated water. Continuous chert layers perch the shallow groundwater and are responsible for five small springs which lie downslope of the plots (Brahana *et al.*, 2005).

Infiltration measurements and soil physical properties

Figure 6 shows a map of the spatial distribution of bulk density on each of the plots. Mean bulk density of plot 1 ($1{\cdot}17$ g cm^{-3}) was significantly ($P = 0{\cdot}042$) greater than the $1{\cdot}01$ g cm^{-3} computed for plot 3 (Figure 6). Although the mean bulk density of plot 2 was greater than that of plot 3, these differences were not statistically significant at $P = 0{\cdot}05$. Generally, bulk densities decreased from upslope (plot 1) to downslope (plot 3) of the field. The high percentage of chert on plot 1 probably accounted for the greater bulk density. The geophysical and drilling data suggest that the soil–regolith/bedrock contact is much thinner beneath plots 1 and 2 than plot 3. Although the mean bulk densities seem to be on the lower end of that reported by previous work in Basin 1 ($1{\cdot}10$–$1{\cdot}46$ g cm^{-3}; Sauer *et al.*, 1998; Sauer and Logsdon, 2002), the results support the general trend where Nixa cherty silt loam soils have greater bulk densities than Captina silt loams.

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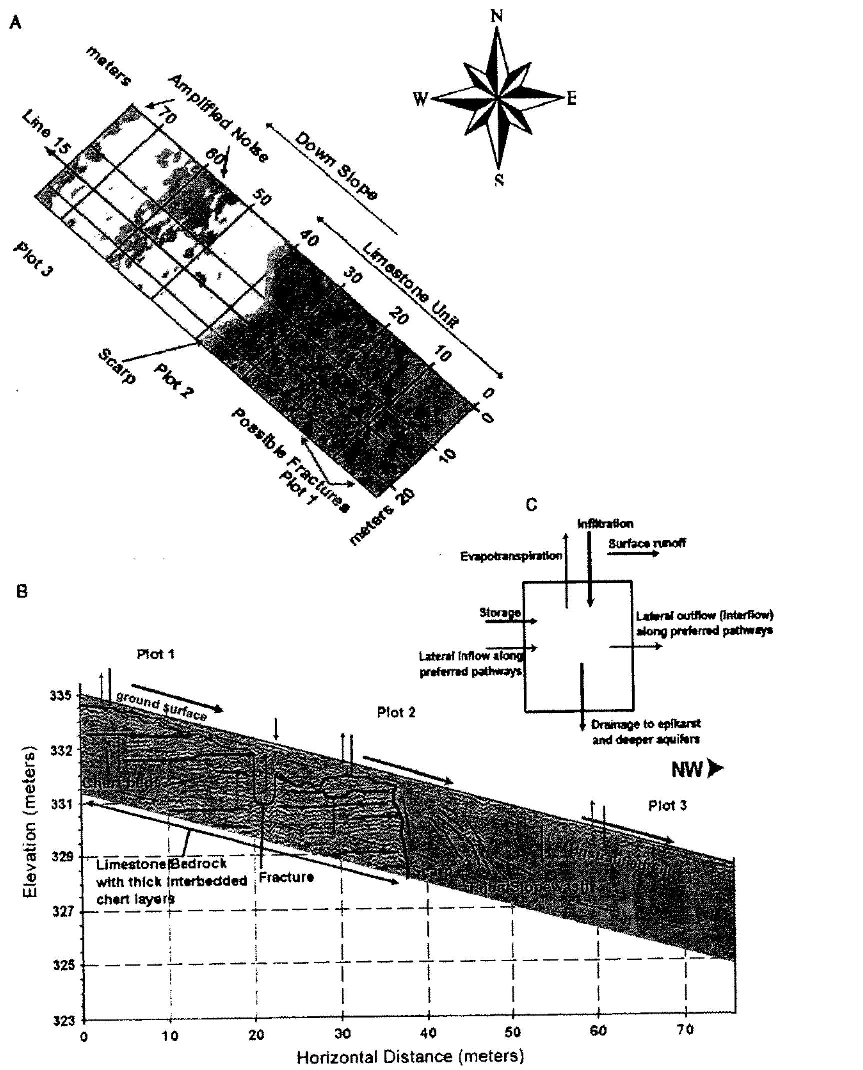

Figure 5. Conceptual model of ground penetrating radar (GPR) measurements showing (A) a composite overlay of GPR data at 12 different depths, location of transect 15, (B) details of GPR transect 15 (line 15) along with flow pathways for water, and (C) a water budget at the Savoy Experimental Watershed study plots (modified from Ernenwein and Kvamme, 2004)

Mean infiltration rate on plot 1 (302 mm h^{-1}) was significantly ($P < 0.0082$) greater than that of plots 2 and 3 (214 mm h^{-1}). There was no significant difference between plots 2 and 3 infiltration rates (Table I). The greater infiltration rates on plot 1 could be a result of the presence of fractures in the subsurface layer of the soil, which is consistent with the resistivity and GPR surveys. These fractures could serve as macropores and induce preferential flow. Ponded infiltration rates were highly variable when compared with measurements made by other researchers on similar soil types. Infiltration rates were between ten and fifteen times greater than that reported by Sauer *et al.* (1998) (20·4 and 22·2 mm h^{-1} for Nixa and Clarksville soils respectively), between four and six times greater than values reported by Sauer *et al.* (2000) (49·0 and 54·4 mm h^{-1} for Nixa and Clarksville soils respectively) and between two and one-half times greater than values reported by Sauer and Logsdon (2002) (158 and 139 mm h^{-1} for Nixa and Clarksville soils respectively) using different infiltration

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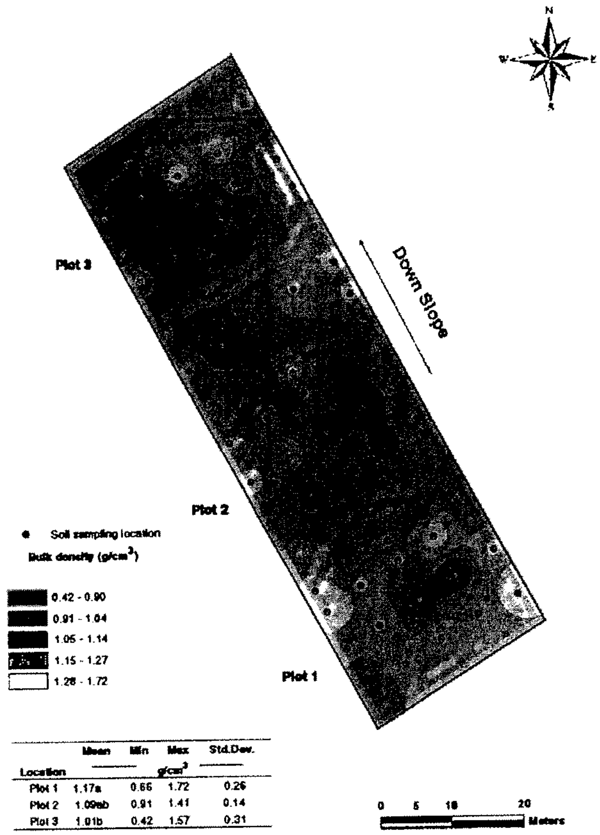

Location	Mean	Min	Max	Std.Dev.
	g/cm³			
Plot 1	1.17a	0.66	1.72	0.26
Plot 2	1.09ab	0.91	1.41	0.14
Plot 3	1.01b	0.42	1.57	0.31

Figure 6. Spatial distribution of bulk density over study plots. Mean values followed by the same letters are not significantly different at $P = 0{\cdot}05$ as determined by Fisher's protected Least Significant Difference method

measuring techniques at a site about 0·25 km away from our study plots. Estimated infiltration rates during rainfall simulations from Haggard *et al.* (2005a) varied between 83 mm h^{-1} and 103 mm h^{-1} at small plots on a Captina silt loam. Other plot studies have observed infiltration rates of approximately 45 mm h^{-1} on a Captina silt loam variable slope box (Haggard *et al.*, 2005b). The current measured infiltration rates were on the higher side of those reported. The variability in the infiltrations rates among the different studies of similar soil types highlights the complex nature of the infiltration–runoff process in Basin 1.

Although the maximum 5 min rainfall intensities during the 1–5 January 2005 events (Table II) did not exceed the measured infiltration rates, a large number of occurrences of the infiltration excess mechanism were observed. It is possible that the variability of infiltration rates across the plots was not adequately represented since the infiltration measurements were performed at only a few (three per plot) locations on the plots. A further contribution to the high infiltration rates could be the locations of bedrock fractures underlying the subsurface soil profile (Figure 5A), or subtle elevation changes (centimetre scale) in the epikarst surface. Although not observed during this testing period, insect burrows are widespread on the plots during part of the year (John Murdoch, written communication, 2005); infiltration processes elsewhere have been documented as

Table I. Summary of infiltration measurements and estimated saturated hydraulic conductivity performed on each plot

Location	Infiltration Rate (mm/hr)		Saturated Hydraulic Conductivity (mm/hr)	
	Mean	Std Dev.	Mean	Std Dev.
Plot 1	302a*	23	173a	13
Plot 2	214b	23	116b	12
Plot 3	214b	23	120b	12

* Mean values within a column followed by the same letter are not significantly different at P = 0·05 as determined by Fisher's protected least significant difference method.

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being dominated by macropores (Bronstert, 1999). The measurement locations were randomly chosen based on the most suitable flat surface, and the heterogeneous and anisotropic response to flow in these plots is consistent with preferred flow pathways.

Estimated saturated hydraulic conductivity on plot 1 (Table I) was significantly greater than that of plots 2 and 3 ($P < 0.0036$). No significant difference was observed between plots 2 and 3 hydraulic conductivities. While measured saturated hydraulic conductivities were lower than values (296 mm h^{-1} for Captina soils and 312 mm h^{-1} for Nixa soils) reported by Sauer *et al.* (1998), they were between one-half and two times greater than values reported (80·2 mm h^{-1} for Nixa and 89·3 mm h^{-1} for Clarksville soils) in the Sauer and Logsdon (2002) study.

Runoff mechanisms on the plots

Both infiltration excess and saturation excess runoff occurred intermittently at the study site from April 2004 through calendar year 2005. Overall, the predominant runoff mechanism was infiltration excess, covering about 58% of the total area, and was somewhat confined to the upper portions of the hillslope (plots 1 and 2) (Figure 7). Saturation excess runoff occurs at about 26% of the total field area and appeared to be dominant on plot 3 and along the south-western boundary of plot 1. Many other researchers have documented the occurrence of saturation excess runoff near streams and at the lower portions of a hillslope (Hewlett, 1961; Dunne and Black, 1970a, 1970b; Srinivasan *et al.*, 2002; Needleman, 2002; Rezzoug *et al.*, 2005; Badoux *et al.*, 2006). 16% of the field did not contribute to significant runoff over the period of study. It must be noted here that saturation excess runoff does occur briefly at other locations of the plots during and between storms and has a greater likelihood of occurrence during long-duration storm events.

Table II. Summary of selected rainfall characteristics and infiltration excess runoff events for 1–5 January 2005 storm events (maximum rainfall intensities in bold)

Date	Time (hours)	5-min Rainfall Intensity (cm/min)	Cummulative Rainfall (cm)	No. of 5-min Infiltration Excess Events
1/1/05	5:05 PM	<0·01	0·03	0
1/2/05	2:40 PM	0·014	0·17	3
1/2/05	2:45 PM	**0·040**	0·39	0
1/2/05	2:55 PM	0·022	0·57	2
1/3/05	2:10 AM	0·003	0·18	2
1/3/05	2:40 AM	0·020	0·47	2
1/3/05	4:30 AM	**0·027**	2·04	3
1/3/05	4:35 AM	**0·027**	2·17	2
1/3/05	4:55 AM	0·024	2·59	2
1/4/05	2:00 AM	0·010	0·08	2
1/4/05	2:40 AM	0·020	0·30	2
1/4/05	5:30 AM	0·041	1·78	8
1/4/05	5:35 AM	0·030	1·93	8
1/4/05	6:15 AM	**0·046**	2·97	15
1/5/05	5:20 AM	0·015	0·76	7
1/5/05	5:30 AM	0·010	0·84	10
1/5/05	6:30 AM	**0·025**	1·50	11

Detailed rainfall–runoff data for events that occurred from 1 January 2005 to 5 January 2005 are shown in Table III. The flow measured at the flume of plot 1 is compared with the location of runoff mechanism occurrence at different times in Table IV. Plot 1 was obviously dominated by infiltration excess, and sensor 106 (see Figure 2 for location) indicated the occurrence of saturation excess. The subsurface material could be a possible explanation for this. The presence of fractures at that location could serve as a reservoir which, when filled up, could lead to the initiation of saturation excess flow.

At 3:00 am, only sensor 104 and 106 indicated runoff. The absence of flow at the flume downslope could only suggest that the surface runoff reinfiltrated before reaching the flume. However, at 6:00 am, six sensors indicated the presence of runoff, suggesting that at least some of the runoff was able to flow downslope in order to be captured by the H-flume. A similar pattern could be observed at 10:00 am, 1:00 pm and 10:00 pm, where no flow was observed at the H-flume but the occurrence of runoff indicates reinfiltration of the runoff. Conversely, flow and corresponding runoff indication at 7:00 am, 8:00 pm and 11:00 pm suggests the runoff was able to flow to the flume via an overland flow route.

Runoff-contributing areas

The rainfall events of 1–3 January 2005 occurred after low-flow (dry) conditions were present beneath the plots (Figure 11). For a 3-week period (17–31 December 2004) before the storm of 2 January, no precipitation was recorded, except for trace amounts of snow dustings on 21 and 22 December, and base-flow recession was nearly flat. After more than 4 cm of rainfall in a period of about 48 h (Figure 3), the storms of 4/5 January 2005 (Figure 3) could be considered to have occurred under wet watershed conditions. The rainfall events of the previous days had raised the level of the groundwater table by as much as 60 to 70 cm (Figure 11) and provided well saturated soil moisture conditions. This was confirmed visually by field visits to the study site during the storm period, and are discussed in detail below.

Table III. Rainfall–runoff data for 1–5 January 2005

Date	Cumulative Rainfall (cm)		Cumulative Runoff (cm)		
	Plots*	Weather Station	Plot 1	Plot 2	Plot 3
1/1/05	0·03	0·08	0	0	0
1/2/05	0·91	0·89	0·03	<0·01	<0·01
1/3/05	3·66	3·63	0·03	0·18	0·05
1/4/05	5·39	4·83	0·13	0·69	0·13
1/5/05	3·18	2·92	0·18	0·15	0·1
5-day Total	13·16	12·34	0·37	1·02	0·28

* Mean total rainfall observed on plots 1 through 3.

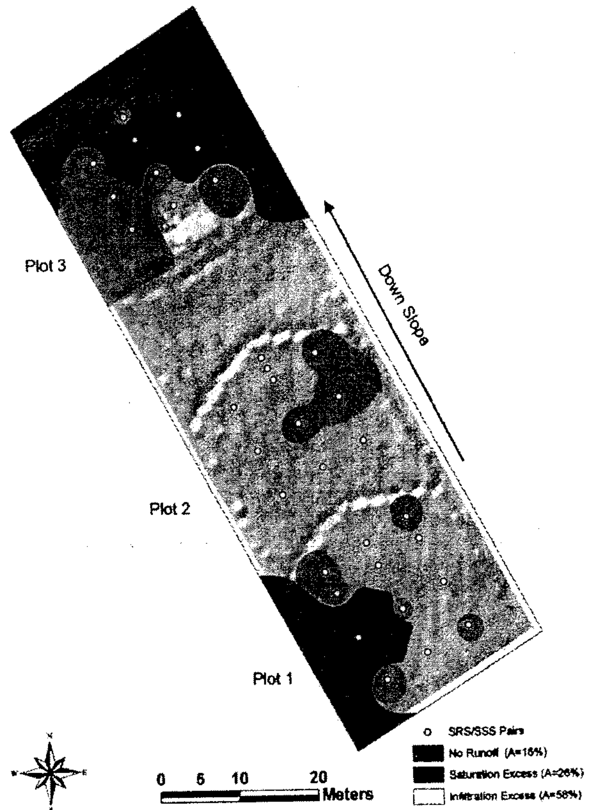

Figure 7. Map of dominant runoff process that occur at study site, Savoy, Arkansas. A is the total percentage area of a runoff process over the area of the three plots

Table IV. Flow and runoff mechanisms observed on plot 1 for 4 January 2005 event (N indicates no runoff mechanism, SE indicates saturation excess and IE indicates infiltration excess runoff)

Time (hours)	Flow measured at H-flume (m^3 s^{-1})	Sensor ID (See Figure 2 for location of sensors in plot)											
		101	102	103	104	105	106	107	108	109	110	111	112
3:00 AM	0	N	N	N	IE	N	SE	N	N	N	N	N	N
6:00 AM	1·20E-04	N	IE	N	IE	N	SE	N	IE	IE	N	IE	N
7:00 AM	7·00E-05	N	IE	N	IE	N	SE	N	IE	IE	N	IE	N
10:00 AM	0	N	N	N	IE	N	SE	N	IE	N	N	IE	N
1:00 PM	0	N	N	N	IE	N	SE	N	N	N	N	IE	N
8:00 PM	1·40E-05	N	N	N	IE	N	SE	N	N	IE	N	IE	N
10:00 PM	0	N	N	N	IE	N	SE	N	IE	N	N	IE	N
11:00 PM	5·20E-06	N	N	N	IE	N	SE	N	IE	N	N	IE	N

Runoff-contributing areas during dry watershed conditions. A minor rainfall event that occurred on 1 January 2005 was a short-duration low-intensity storm that totaled 0·03 cm (Table II, Figure 3). None of the runoff sensors responded to this event (Figure 8), nor was springflow affected (Figure 12). The 2 January event started at 2:30 pm, by 2:35 pm, three sensors had responded to the presence of runoff and 1·5% of the total area (plots 1–3) contributed to runoff (Figure 8). Rainfall intensity within this period was 0·014 cm min^{-1} and the runoff process was infiltration excess. After 10 min of the rainfall event the total infiltration excess area increased to 4%. The

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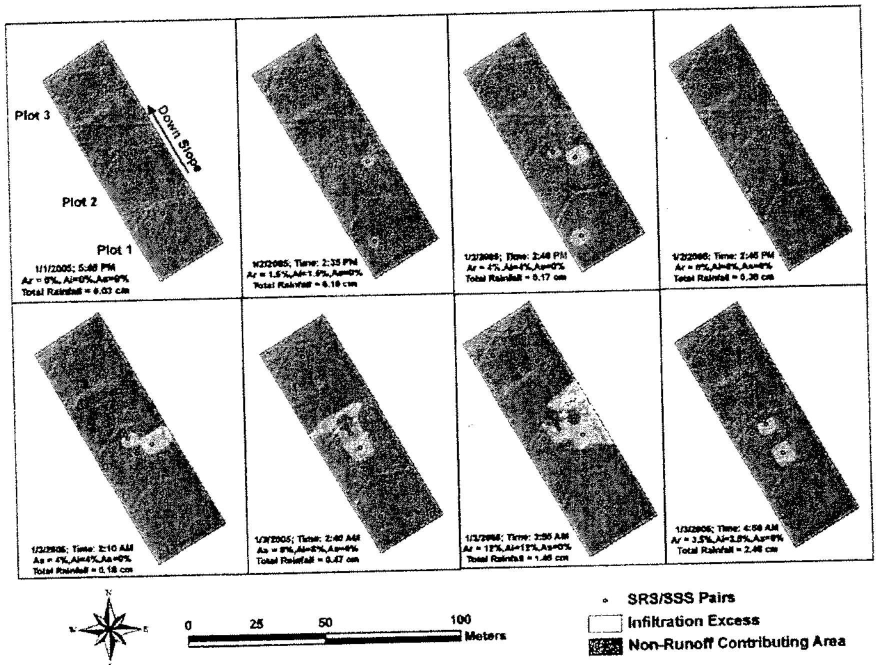

Figure 8. Map of runoff contributing areas on each plot for 1–3 January 2005 storm events. A_r is the total percentage runoff contributing area, A_i is the percentage of infiltration excess runoff area, and A_s is the percentage of saturation excess runoff area computed over all the plots

storm attained the highest intensity of 0·04 cm min^{-1} at 2:45 pm (Table II), but surprisingly, none of the sensors indicated the occurrence of runoff (Figure 8). Within this 5 min period, the initial runoff areas contracted, and the runoff at these sensors reinfiltrated into the soil profile; quite possibly this may be due to the development of a matric suction gradient in the relatively dry soil layers below after the soil surface had been subjected to initially wet conditions. The rainfall event ended at 4:15 pm with a cumulative total of 0·91 cm (Figure 3). The 3 January event started at 12:25 am with an intensity of 0·002 cm min^{-1}. By 2:10 am, the cumulative rainfall was 0·18 cm and 4% of the plots were contributing to infiltration excess runoff (Figure 8). At 2:40 am, the runoff contributing area had increased by 4%. The maximum infiltration excess runoff area (12%) occurred at 3:55 am with a cumulative rainfall of 1·45 cm and an intensity of 0·02 cm min^{-1}. The runoff-contributing areas decreased to 3·5% by 4:50 am with 2·48 cm total rainfall. The rainfall event ended at 5:55 am with a cumulative total of 3·66 cm (Table II). The absence of saturation excess runoff areas during the 1–3 January rainfall events (Figure 8) was consistent with the theory of saturation excess runoff development. The groundwater wells on each of the plots indicated water levels below the ground surface for most of the time until well 1 rose to the ground surface during the 3 January storm event. Owing to the limited spatial extent of the infiltration measurements, no conclusions can be drawn about the effects of infiltration rates; nonetheless, it is quite obvious that the higher bulk densities observed on plots 1 and 2 played a major role in the infiltration excess process during these storm events. Overall, the 1–3 January rainfall events resulted in runoff-contributing areas that were somewhat confined to plot 2 and consistent with the cumulative runoff depth that was observed on the plot (Figure 8, Table III).

Runoff-contributing areas during wet watershed conditions. The 4 January 2005 rainfall event started at 2:00 am (Figure 3). By 3:00 am, 0·33 cm of rainfall had fallen and three sensor locations indicated the occurrence of infiltration excess runoff (Figure 9). At 6·00 am 3 h later, the extent of the runoff-contributing area had increased but for the most part was confined to plots 1 and 2 (Figure 9). The groundwater well on plots 1 and 2 indicated water levels at the ground surface, while well 3 continued to rise (Figure 11) at this time. Most of the runoff (20%) produced was infiltration excess whereas only 6% of the runoff area was the result of the saturation excess mechanism. A possible explanation for the observed saturation excess areas could be macropore storage of flows in these areas (Figure 5). After a total rainfall of 3·23 cm (7:00 am), the maximum runoff extent (51%) was observed, 27% was due to infiltration excess and 24% was due to saturation excess mechanism. During this period, well 3 was observed to briefly rise to

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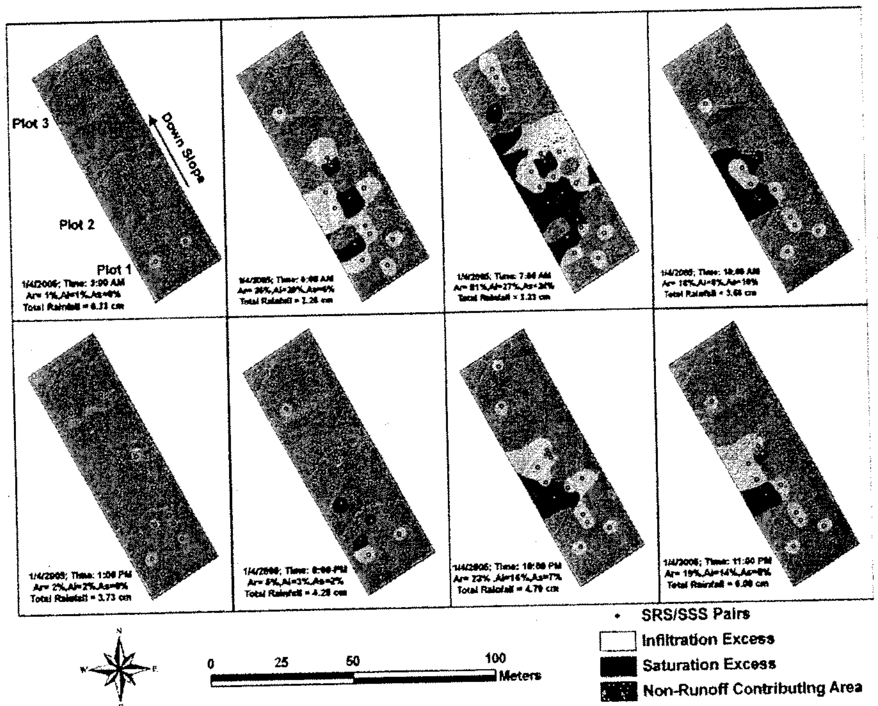

Figure 9. Map of runoff-contributing areas on each plot for 4 January 2005 storm event. A_r is the total percentage runoff contributing area, A_i is the percentage of infiltration excess runoff area, and A_s is the percentage of saturation excess runoff area computed over all the plots

ground surface. The expanded saturation excess area suggests increased storage areas in the macropores. At 10:00 am, the saturation excess process seemed to dominate the runoff generating process, occurring in 10% of the area whereas infiltration excess occurred in 8% of the area. The runoff areas were seen to contract to just a few spots and reverted to pure infiltration excess (2% contribution) around 1:00 pm. Another burst of rainfall activity, which started around 7:00 pm, caused the contributing areas to respond with expansion ($A_r = 23\%$, at 10:00 pm) and contraction ($A_r = 19\%$ at 11:00 pm) until midnight. Infiltration excess was the dominant runoff mechanism at both times. Both events contributed to a rainfall total of 5·39 cm at the end of the day (Table III).

The 5 January 2005 event (Figure 10) was a continuation of the 4 January late evening storm. The sensors were observed to respond to runoff by the end of the first hour (1:00 am) with a total runoff area of 5%, most of the runoff process being infiltration excess (4%). During that time, 0·08 cm of rainfall occurred. Two hours and 0·28 cm of rainfall later, the runoff-contributing area had increased marginally. By 6:00 am, a total of 1·14 cm of rain had fallen and the runoff-contributing area expanded further (Figure 10). The largest extent of runoff contribution occurred at 7:00 am. Within that hour, 0·64 cm of rainfall had occurred. As the rainfall amount decreased, the runoff-contributing area also decreased and by 3:00 pm, only the south-western part of plot 1 and south-eastern part of plot 3 contributed to runoff. Rainfall total for the storm from midnight 5 January to midnight 6 January was 3·18 cm (Figure 3). Total runoff depth was 0·18 cm for plot 1, 0·15 cm for plot 2, and 0·10 cm for plot 3. Runoff sensor 215 on plot 2 (Figure 2) indicated saturation excess runoff throughout the storm event (Figure 10). The GPR data is in conceptual agreement with this finding. The underlying chert layers have been fractured (Figure 5A) and the weathered bedrock surface of the epikarst is locally irregular with highs and lows (Figure 5B). If the subsurface drainage (Figure 5C) below areas where groundwater is perched or plugged cannot be transported rapidly, soil pores and fractures remain saturated, resulting in a saturation excess condition such as sensor 215 on plot 2 (Figure 10). The extent of the saturation excess area then depends on the amount of infiltrated water supplied to the area.

In contrast to the 4 January storm event, the 5 January storm resulted in a greater extent of saturation excess area on plot 3. This may be attributed to lateral subsurface flow from upgradient plots 1 and 2 along preferred pathways to plot 3 (Figure 5), enhancing the saturation excess generating process on plot 3.

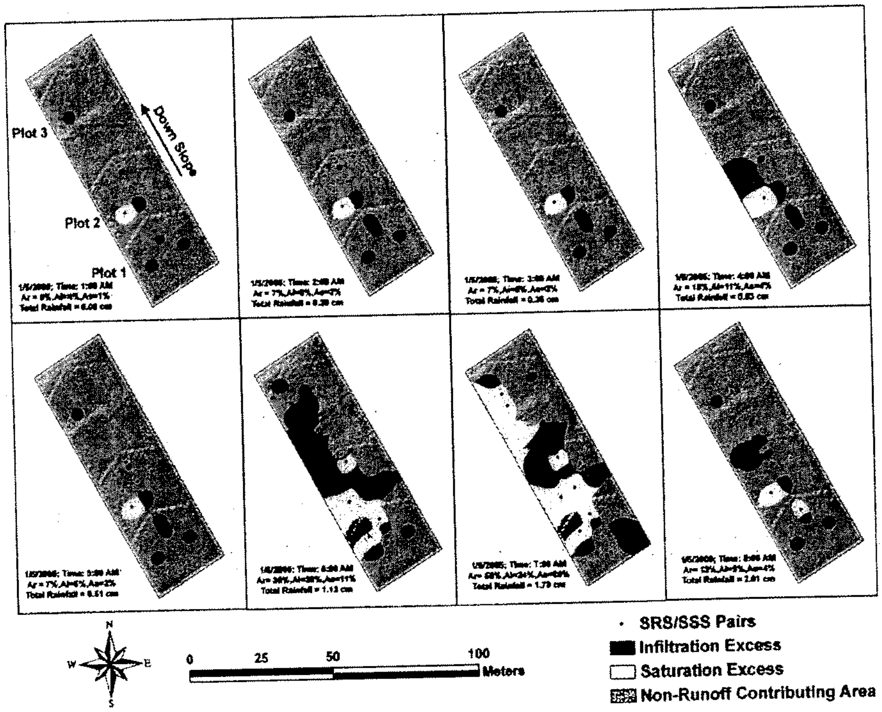

Figure 10. Map of runoff contributing areas on each plot for 5 January 2005 storm event. A_r is the total percentage runoff contributing area, A_i is the percentage of infiltration excess runoff area, and A_s is the percentage of saturation excess runoff area computed over all the plots

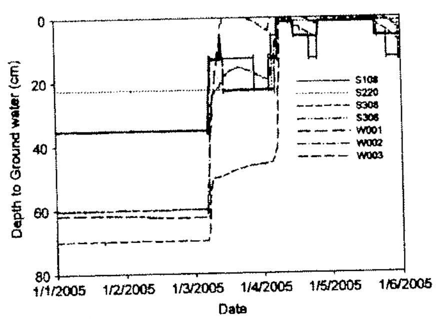

Figure 11. Water table dynamics for period 1–5 January 1 2005 as indicated by monitoring wells and nearest subsurface sensors. 'S' denotes subsurface sensors and 'W' denotes wells. Locations of subsurface sensors and monitoring wells are shown in Figure 2

Water table dynamics and flow responses

Figure 11 shows a graph of the response of the shallow monitoring wells on each plot with their corresponding nearest subsurface sensors. The subsurface sensor and monitoring well data were significantly correlated ($R^2 =$ 0·90, 0·67, 0·96 on plots 1, 2, and 3, respectively, $P <$ 0·05) and were able to capture the subsurface dynamics of the water table response to the precipitation. The time of occurrence of the relative expansion and contraction of the runoff source areas were coincident with the rise and fall of water table for the corresponding rainfall totals (Figures 3 and 8–11).

The spring flow and runoff responses to rainfall before, during, and after the 4/5 January 2005 events are shown in Figure 12. A lag in response of the spring during the 3 January storm event compared to the almost immediate response time during the 4 January storm event can be seen from this figure. Further examination of the spring hydrographs for different rainfall events revealed that the lag in spring response to rainfall is highly variable depending on rainfall intensity and antecedent soil moisture characteristics. For example, during the 2 January storm event, the spring showed a slight response 30 min after the start of rainfall but quickly receded to base flow within 15 min. For the 3 January rainfall event, spring response was observed approximately 3 h after the start of the rainfall event. However, the spring was observed to respond to the 4 January 2005 rainfall event approximately 40 min after the start of rainfall.

Examination of the groundwater wells during the dry watershed conditions is enlightening, in that the wells in plots 1 and 2 indicated low groundwater levels until the 3 January storm when they rose rapidly in response to infiltration, and after the most intense part of the storm,

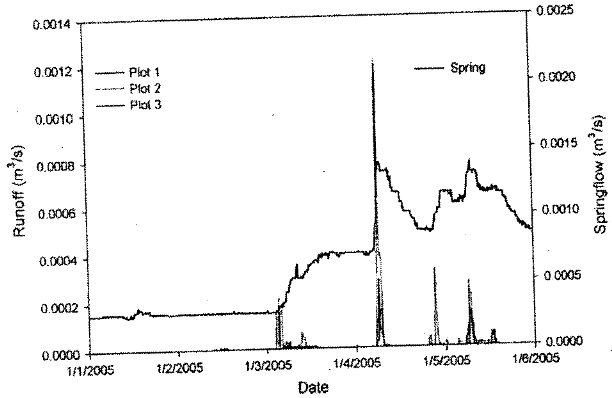

Figure 12. Spring flow and runoff on each plot for period for period 1–5 January 2005

peaked and then went into recession. The well in plot 3, however, continued rising throughout that period, until infiltration from the next storm caused a rapid rise of all wells to near ground surface the next day (Figure 11). This continual rise in well 3 is interpreted to be caused by interflow from upgradient plots 1 and 2 moving laterally along permeability contrasts in the soil zone. This lateral flow allows recession in wells 1 and 2, and continued rise in well 3. Conceptually, this is represented by dry condition flow arrows in Figure 5. Insofar as groundwater level in well 3 starts receding more steeply than the water level in well 2, it would appear that downward vertical drainage in plot 3 is more effective (more well connected) than plots 1 and 2. The wells were completed by augering to refusal, and the sequence of lithologies encountered is very similar. Thinner regolith and soil in plots 1 and 2, perched lateral interflow following the epikarst, and enhanced underdraining to the St. Joe Limestone via faults and joints represents a consistent water budget for the area, which explains most of the observed data.

For wet watershed conditions, well 1 remains nearly completely saturated after the second major pulse of infiltration through the remainder of the period of record (Figure 11). Water level rises and recessions are rapid, but the thickness of the aquifer under plot 1 between land surface and the top of the perching chert seems to be less, an observation consistent with the geophysical data and well drilling (Figure 5). Well 2 is intermediate in its response, both in timing and in magnitude, and well 3 shows the greatest transmissivity and storativity of the plots. Insofar as there is no justification that hydraulic conductivity and specific storage vary predictably over the plots, the thickness of the unconfined, porous media of the shallowest aquifer most likely is simply a result of greater thickness moving from divide to drain. Coupled with enhanced underdraining beneath fractures and faults (Figure 5B), the piling up of water in plot 2 seems like a rational consequence of interflow along preferred flowpaths and underdraining along joints and faults.

SUMMARY AND CONCLUSIONS

A field-scale methodology was used to identify and delineate runoff mechanisms and critical runoff-contributing areas in a hillslope in the Savoy Experimental Watershed. Results from this study showed that both infiltration excess and saturation excess runoff processes occur on this hillslope. Although variability of infiltration rates across the plots may not have been adequately represented, the trends in soil hydraulic properties and the subsurface attributes of the plots were instrumental in locating runoff processes in the field. The infiltration excess runoff mechanism areas were located primarily in areas of high soil electrical resistance while saturation excess mechanism areas were located in areas of subsurface fractures, low soil electrical resistance, relatively homogenous subsurface soil material, and on the downslope end of the field. The contracting and expanding dynamics of runoff generating areas were shown for five rainfall events that occurred in January 2005. Given that very little data exist on hydrologic processes and their interactions with runoff contributing areas, further study is required to develop a thorough understanding of this area of hydrology. This methodology provides a detailed procedure for capturing the hydrologic activities that occur on a hillslope and provides benchmark procedures that can be used in locating areas for best management practice (BMP) implementation.

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REFERENCES

Anderson MG, Burt TP. 1978. Toward more detailed field monitoring of variable source areas. *Water Resources Research.* **14**: 1123–1131.

Badoux A, Witzig J, Germann PF, Kienholz H, Lüscher P, Weingartner R, Hegg C. 2006. Investigations on the runoff generation at the profile and plot scales, Swiss Emmental. *Hydrological Processes* **2**: 377–394. DOI: 10·1002/hyp.6056.

Betson RP. 1964. What is watershed runoff? *Journal of Geophysical Research.* **69**: 1541–1552.

Brahana JV, Ting TE, Al-Qinna M, Murdoch JF, Davis RK, Laincz J, Killingbeck JJ, Szilvagyi E, Doheny-Skubic M, Chaubey I, Hays PD, Thoma G. 2005. Quantification of hydrologic budget parameters for the vadose zone and epikarst in mantled karst: In *Proceedings of the U.S. Geological Survey Karst Interest Group*, Rapid City, South Dakota, September 12–15, 2005, Kuniansky EL (ed). US Geological Survey Scientific Investigations, Report 2005–5160; 144–152.

Bronstert A. 1999. Capabilities and limitations of detailed hillslope hydrological modeling. *Hydrological Processes* **13**: 21–48.

Dunne T, Black RD. 1970a. An experimental investigation of runoff production in permeable soils. *Water Resources Research* **6**: 478–490.

Dunne T, Black RD. 1970b. Partial area contributions to storm runoff in a small New England watershed. *Water Resources Research.* **6**: 1296–1311.

Dunne T, Moore TR, Taylor CH. 1975. Recognition and prediction of runoff-producing zones in humid regions. *Hydrologic Sciences Bulletin* **20**: 305–327.

Ernenwein EG, Kvamme KL. 2004. Geophysical Investigations for subsurface fracture detection in the Savoy Experimental Watershed, Arkansas. Archeo-Imaging Lab unnumbered report, University of Arkansas, Fayetteville, AR, 1–48.

Freeze ET. 1974. Streamflow generation. *Reviews of Geophysics and Space Physics* **12**: 627–647.

Gburek WJ, Sharpley AN. 1998. Hydrologic controls on phosphorous loss from upland agricultural watersheds. *Journal of Environmental Quality* **27**: 267–277.

Haggard BE, DeLaune PB, Smith DR, Moore Jr PA. 2005a. Nutrient and b17-estradiol loss in runoff from various poultry litters. *Journal of American Water Resources Association* **41**: 245–256.

Haggard BE, Moore Jr PA, Brye KR. 2005b. Effect of slope on runoff from a small variable slope box-plot. *Journal of Environmental Hydrology* **13**: Paper No. 25.

Harper MD, Phillips WW, Haley GJ. 1969. Soil survey of Washington County, Arkansas. US. Department of Agriculture, Soil Conservation Service, US Government Printing Office, Washington, DC.

Hewlett JD. 1961. Soil moisture as a source of base flow seep mountain watersheds. Southeast Forest Experiment Station Paper 132. US Forest Service, US Government Printing Office, Washington, DC.

Hewlett JD, Hibbert AR. 1967. Factors affecting the response of small watersheds to precipitation in humid regions. In *Forest Hydrology*, Sopper WE, Lull HW (eds). Pergamon Press: Oxford; 275–290.

Holden J, Burt TP. 2003. Runoff production in peat covered blankets. *Water Resources Research* **39**: 1191. DOI:10·1029/2002WR001956.

Horton RE. 1933. The role of infiltration in the hydrologic cycle. *Transaction of the American Geophysical Union* **14**: 446–460.

Needleman BA. 2002. *Surface runoff hydrology and phosphorus transport along two agricultural hillslopes with contrasting soils*. PhD thesis. The Pennsylvania State University, University Park, PA.

NOAA. 2002. Climatography of the United States No. 81. Monthly station normals of temperature, precipitation, and heating and cooling degree days 1971–2000: Arkansas. Asheville, NC.

Pionke HB, Gburek WJ, Sharpley AN. 1997. Hydrologic and chemical controls on phosphorous loss from catchments. In *Phosphorous loss from soil to water*, Tunney H, Carton OT, Brookes PC, Johnston AE (eds). CAB International Press: Cambridge; 225–242.

Rezzoug A, Schumann A, Chifflard P, Zepp H. 2005. Field measurement of soil moisture dynamics and numerical simulation using the kinematic wave approximation. *Advances in Water Resources* **28**: 917–926.

Sauer TJ, Moore Jr PA, Coffeey KP, Rutledge EM. 1998. Characterizing the surface properties of soils at varying landscape positions in the Ozark highlands. *Soil Science* **163**: 907–915.

Sauer TJ, Daniel TC, Nichols DJ, West CP, Moore Jr PA, Wheeler GL. 2000. Runoff water quality from poultry litter-treated pasture and forest sites. *Journal of Environmental Quality* **29**: 515–521.

Sauer TJ, Logsdon SD. 2002. Hydraulic and physical properties of stony soils in a small watershed. *Soil Science Society of America Journal* **66**: 1947–1956.

Schindler DW, Armstrong FAJ, Holmgren SK, Brunskill GJ. 1971. Eutrophication of Lake 277, Experimental Lakes Area, Northwestern Ontario, by addition of phosphate and nitrate. *Journal of the Fisheries Research Board of Canada* **28**: 1763–1782.

Soerens TS, Fite EH III, Hipp J. 2003. Water quality in the Illinois River: Conflict and cooperation between Oklahoma and Arkansas. In *Proceedings of Diffuse Pollution Conference*, International Water Association; 14–19.

Srinivasan MS, Wittman MA, Hamlett JM, Gburek WJ. 2000. Surface and subsurface sensors to record variable runoff generation areas. *Transactions of the American Society of Agricultural Engineers* **43**: 651–660.

Srinivasan MS, Gburek WJ, Hamlett JM. 2002. Dynamics of storm flow generation-A hillslope scale field study in east-central Pennsylvania, USA. *Hydrologic Processes* **16**: 649–665.

Walter MT, Walter MF, Brooks ES, Steenhuis TS, Boll J, Weiler KR. 2000. Hydrologically sensitive areas: Variable source area hydrology implications for water quality risk assessment. *Journal of Soil and Water Conservation* **3**: 277–284.

Zollweg JA. 1996. Field study to support hydrologic modeling and analysis of watershed function at the microscale. In *Proceedings of Watershed Restoration Management—Physical, Chemical, and Biological Consideration.* American Water Resources Association; 129–134.

Zollweg JA, Gburek WJ, Pionke HB, Sharpley AN. 1995. GIS- based delineation of source areas of phosphorous within agricultural watersheds of the northeastern USA. In *Proceedings of a Boulder Symposium on Modeling and Management of Sustainable Basin-scale Water Resource Systems.* International Association of Hydrological Sciences; 31–39.

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Nitrogen and Phosphorus Concentrations and Export from an Ozark Plateau Catchment in the United States

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In 1993–1995, the Beaver Lake Basin contained about 2000 poultry houses producing about 200 000 Mg yr^{-1} of poultry litter, and 8000 and 4000 Mg yr^{-1} of nitrogen (N) and phosphorus (P), respectively. Most of the poultry litter was land applied as a fertiliser to meet forage N requirements, making it susceptible to transport from the landscape during episodic precipitation events. Nitrogen and P concentrations were measured in four sub-watersheds of Beaver Lake, a reservoir on the White River in Arkansas, USA, to assess possible relationships between pasture land use and stream nutrient concentrations and export. Surface water samples were collected 17 times annually for 2 years from ten total stream sites within the four watersheds. Samples were analysed for soluble reactive P (SRP), total P (TP), ammonium-N (NH_4-N), nitrate-N (NO_3-N), total Kjeldhal N (TKN) and total N (TN). Discharge was measured at four gauged stream stations, and nutrient export was calculated using the US Geological Survey ESTIMATOR software and non-biased re-transformation from log space. Stream SRP, NO_3-N and TN concentrations (geometric-mean) increased linearly with per cent of pasture in watersheds, whereas N and P export coefficients increased exponentially with pasture land use. Nutrient export (kg yr^{-1}) increased with basin size, but nutrient yield (kg km^{-2} yr^{-1}) decreased with basin size. Nutrient yield was from three times to over 10 times greater than nutrient yields observed in regional undeveloped streams and the average of the Hydrologic Benchmark Network of the US Geological Survey. It is apparent that pasturelands in this basin affect stream nutrient concentrations and export to Beaver Lake and its tributaries. This investigation emphasises the need to carefully manage poultry litter because small losses of nutrients compared to the total amount of nutrients produced in a basin may still impact stream nutrient concentrations and export.
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Introduction

The growing influence of non-point source impacts on the nation's water quality gained substantial public prominence with the release of US Environmental Protection Agency's 1996 water quality report to Congress, and the 1998 report chronicles similar impairments. Nitrogen (N) and phosphorus (P) loading to rivers and streams often limits the aesthetic value of affected water bodies and the goods and services these ecosystems provide (National Research Council, 1992; Carpenter *et al.*, 1998). Non-point sources may be responsible for >90% of the P load of about one-third of US rivers and streams (Newman, 1996). Drivers of non-point source (NPS) pollution vary regionally, reflecting a combination of land uses, climate, and edaphic conditions; however, it is clear that agricultural land use is a major contributor to NPS contamination of surface and groundwater (Carpenter *et al.*, 1998).

The issue of NPS nutrient loading has come into sharp focus in the state of Arkansas in the last 10 years due to the rapid growth of the poultry industry. Arkansas has ranked first in the United States in poultry production during the early 1990s, with the production of over one billion broilers each year (Arkansas Agricultural Statistics Service, 1995). This industry creates a substantial amount of waste along with commercial products; each individual broiler produces approximately 1 kg of poultry litter, including manure and bedding, and poultry litter is typically applied as a fertiliser on pastures. In the past, poultry manure has been applied to fields based on N demand of crops, and only recently has animal waste been applied on a P basis. The environmental consequences of N based applications included high loads of P, heavy metals and organic compounds in runoff and eventually in the receiving fresh water ecosystems. From 2·2 to 7·3% of total P (TP) in poultry litter applied to pasture surfaces can be lost in runoff, 80% of which is in the dissolved reactive (*i.e.* biologically available) form (Edwards & Daniel, 1993).

Within the Ozark Plateaus, livestock and poultry waste is recognised as a major source of nutrient loading (Davis & Bell, 1998). Further, annual fertiliser use in the White River basin has increased by 77% for P and by 200% for N between 1965 and 1985 (Alexander & Smith, 1990). Petersen (1992) reported trends of increasing N and P concentrations in streams of northwest Arkansas between 1981 and 1989, often associated with human activity and/or poultry farming. Concern for the potential for NPS loading and eutrophication of regional streams, rivers, and reservoirs has subsequently increased in the past 5 years as well. The objectives of this study were to: (1) determine the relationship between land use and stream N and P concentrations in the watershed; (2) estimate nutrient export from the four catchments; and (3) investigate the relationship between nutrient export and land use in the catchments.

2. Materials and methods

2.1. *Site description*

Beaver Lake was constructed in 1963 and is the first of a series of reservoirs on the White River in the Ozark Plateau of northwestern Arkansas. In addition to the White River, Richland Creek, Brush Creek, War Eagle Creek and several smaller streams supply water to the impoundment (*Fig. 1*). Watershed geology consists of limestone, dolomite, sandstone, shale and chert. The Boone and St Joe Formations contain the major regional aquifer. As water moves rapidly without much natural filtering in these formations, the aquifer is highly susceptible to groundwater nitrate contamination by non-point sources such as septic tanks, poultry houses, fertiliser, and landfills. Land cover is a mixture of urban and suburban developments, hardwood forests and agricultural lands. In 1992, the basin contained over 2000 confined animal operations in an area of about 300 000 ha. *Figure 1* shows the location of the study watershed within Arkansas, drainage network and sub-basins, water-quality monitoring sites, and distribution of land use within the watershed.

Ten locations on Beaver Lake tributaries were chosen as sampling sites (Table 1, *Fig. 1*). For most sampling points, sites were situated at the base of the drainage but also sufficiently upstream of the reservoir so that discharge was not affected by impoundment. Sites included (from east to west): two sampling points in the War Eagle Creek basin (sites 1 and 2); one in the Brush Creek basin (site 3); four sites in the Richland Creek drainage (sites 4–7); one site each at the base of the East Fork (site 8); and West Fork (site 9) of the White River. Site 10 was at the head of the Beaver Lake Reservoir on the White River.

2.2. *Field and laboratory procedures*

All sites were sampled approximately 17 times annually for 2 years from August 1993 to June 1995. Samples for year 1 were collected from August 1993 through July 1994 and samples for year 2 from August 1994 through June 1995. Water samples were collected immediately below the water surface with a horizontal style Alpha sampler from bridge crossings or by grab samples from the stream bank. Approximately 500 m*l* of stream water were collected at each site. A 25 m*l* aliquot was immediately filtered through a 0·45 µm membrane filter and acidified to pH 2 with 6 N HCl. The remaining volume of sample was stored on ice and in the dark for later analysis.

Total Keldjhal N (TKN) and TP digestions were performed on acidified, unfiltered subsamples using H_2SO_4 with K_2SO_4 and HgO as catalysts (EPA, 1983). Total P was determined colorimetrically on digests by the automated split reagent ascorbic acid method (EPA, 1983). The automated salicylate-nitroprusside method (Technicon, 1976) was used to measure TKN. Ammonium-N (NH_4-N) and nitrate-N (NO_2-N $+ NO_3$-N, hereafter referred to as NO_3-N) concentrations were measured on acidified, filtered samples utilising a modified microscale determination method (Sims *et al.*, 1995). Total N (TN) is simply the sum of TKN and NO_3-N. The automated ascorbic acid reduction method (APHA, 1992) was used to determine soluble reactive P (SRP) on the acidified, filtered samples.

2.3. *Data analysis*

Watershed boundaries were delineated using ArcView and 1:24 000 digital elevation model (DEM) data

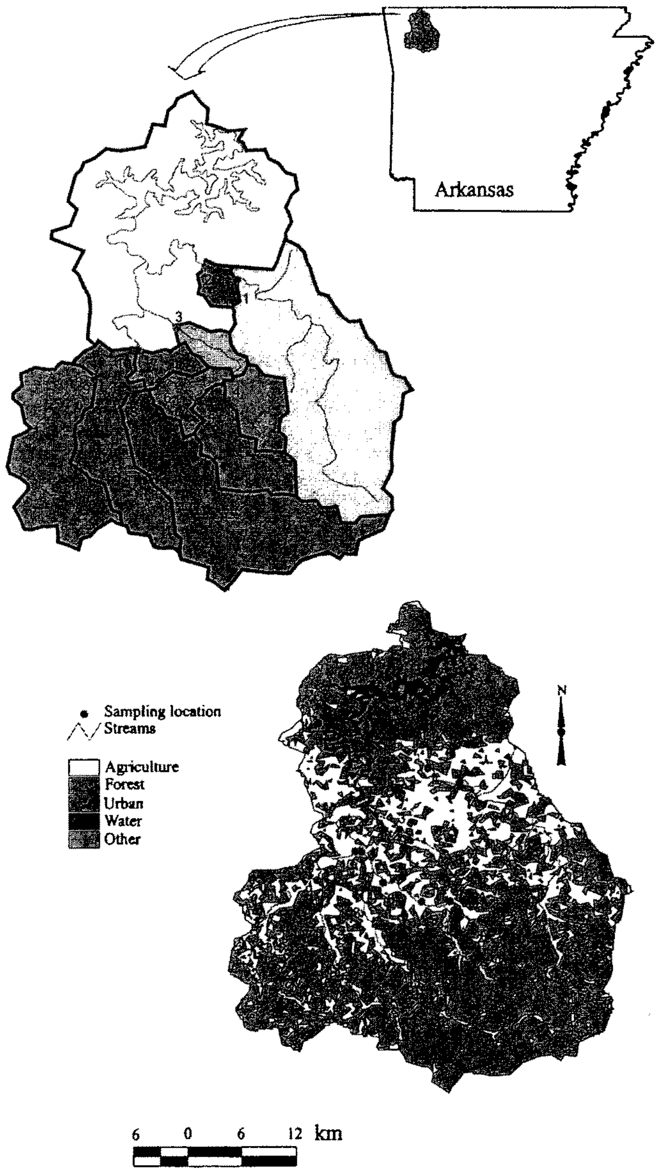

Fig. 1. Location of the Beaver Lake Basin, sampling sites, sub-basins, and the portion of land use categories within the watershed (watershed numbers correspond to numbers in Table 1)

obtained from the US Geological Survey. Water-quality sampling locations were used as the outlets for each sub-basin. Per cent land use for each category and sub-basin was determined using 1992 digital land use – land cover data and ArcView. Simple linear regression was used to assess the effect of land use on water quality by regressing geometric mean nutrient concentrations against the per cent pasture (arcsin transformed; Zar,

Table 1
List of tributary sampling stations within the Beaver Lake Basin, their respective latitude and longitude, sub-basin area, and per cent pasture and forest cover within each sub-basin

Site	*Stream station*	*Sub-basin*	*Latitude*	*Longitude*	*Area, ha*	*Pasture, %*	*Forest, %*
1	Upper War Eagle Creek	War Eagle	36°12′02·4″N	93°50′15·4″W	67 439	37	60
2	Lower War Eagle Creek	War Eagle	36°13′38·6″N	93°54′05·8″W	72 462	38	59
3	Brush Creek	Brush	36°07′57·1″N	93°56′53·6″W	5160	55	42
4	Drake's Creek	Richland	36°01′17·1″N	93°51′36·3″W	4220	31	67
5	Upper Richland Creek	Richland	36°01′09·8″N	93°54′32·3″W	23 985	29	69
6	Middle Richland Creek	Richland	36°02′52·7″N	93°58″30·1″W	30 728	34	64
7	Lower Richland Creek	Richland	36°06′15·3″N	94°00′26·6″W	36 202	37	60
8	West Fork White River	White	36°03′15·1″N	94°04′58·1″W	31 875	31	61
9	East Fork White River	White	36°01′44·0″N	94°01′05·6″W	49 070	16	82
10	White River	White	36°06′21·8″N	94°00′40·8″W	106 408	25	71

1984) in each sub-basin, and the level of significance was set at 0·05. Multiple regression analysis was not used because pasture and forested land uses were strongly related (coefficient of determination, $R^2 = 0{\cdot}98$), and urban land use was 2–4% in all sub-basins except one. Site 10 was excluded from this analysis because upstream point source contributions would interfere with the relationship between land use and nutrient concentrations.

Flow data for War Eagle Creek, Brush Creek, Richland Creek, and White River were obtained from United States Geological Survey stream gauge data, as modified by EGIS (1996). Streamflow was separated into seasonal base flow and storm runoff using a deterministic procedure proposed by the British Institute for Hydrology (1980); the Base Flow Index computer software was developed by Wahl and Wahl (1995) to provide an automated technique for base flow separation. Storm samples were defined as those collected when base flow was less than 70% of total streamflow. Nutrient export coefficients were estimated by log-linear regression of load L and concentration C, discharge Q, time T and seasonal factors (Cohn *et al.*, 1989) using the US Geological Survey ESTIMATOR software program in a concentrations model:

$$\ln(C) = \beta_1 + \beta_2 \ln(Q) + \beta_3 T + \beta_4 \sin(2\pi T) + \beta_5 \cos(2\pi T) \quad (1)$$

and in a load model:

$$\ln(L) = \beta_1 + \beta_2 \ln(Q) + \beta_3 T + \beta_4 \sin(2\pi T) + \beta_5 \cos(2\pi T) \quad (2)$$

where: L is load in kg d^{-1}; C is concentration in mg l^{-1}; T is time in Julian days; and Q is mean daily discharge in m^3 s^{-1}. Simple retransformation from log values was not employed, because this estimator may be biased, and under normal circumstances can underestimate export (Ferguson, 1986); therefore, the ESTIMATOR program implements a minimum variance unbiased estimator (MVUE) described in Cohn *et al.* (1989). Daily loads were estimated and then summed to produce annual loads, and relations between nutrient load, yield and basin characteristics were also evaluated.

The sampling strategy used to estimate nutrient loads in streams also plays a role in the accuracy and precision of the estimates. Robertson and Roerish (1999) suggested that semi-monthly sampling for 2-year studies resulted in not only the least biased but also the most precise estimates when using regression techniques. Furthermore, Green and Haggard (2001) demonstrated that 35 samples collected over a 3-year period provided the information required to accurately estimate nutrient loads via the regression method for high frequency sampling and integration. The streams draining the Beaver Lake Basin were sampled 17 times annually, less than semi-monthly, but greater than or equal to 25% of the samples were collected during storm events. The sampling strategy used for this project adequately represents the range of flow and concentration conditions required producing good estimates of nutrient loads.

3. Results

3.1. *Discharge*

Low flows and few surface runoff events characterised discharge at all sites during summer and autumn (June–October), particularly during 1994, and higher seasonal base flows and more frequent floods characterised discharge during winter and spring (*Fig. 2*). Base flow and numbers and sizes of floods were greater in year 2 at all four gauged sites. Summer base flows were not proportional to basin size, and ranged from 0·2 m^3 s^{-1} at

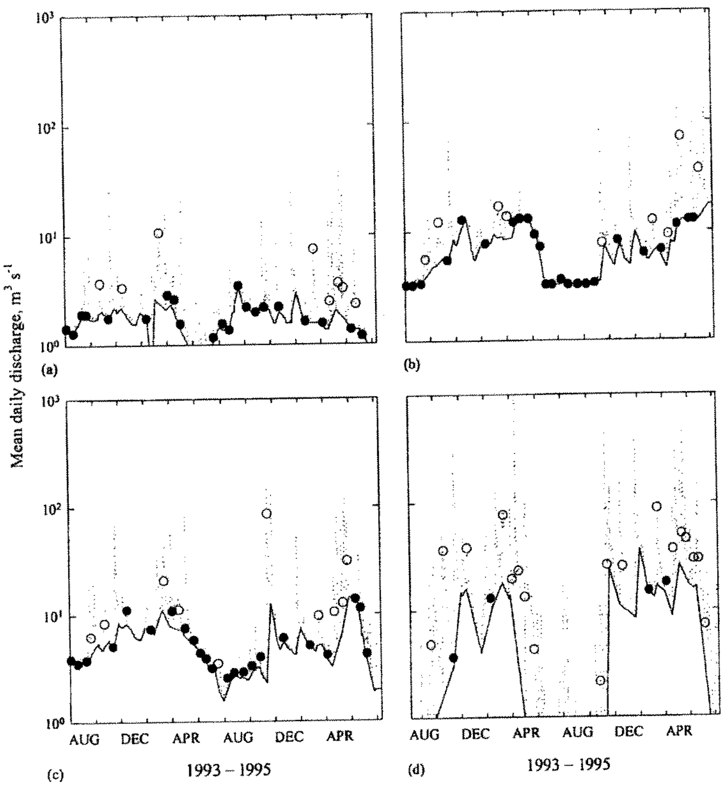

Fig. 2. Sampling date, mean daily discharge and base flow separation at Brush Creek, Richland Creek, War Eagle Creek, and White River during the study period; (a) Brush Creek, site 3; (b) Richland Creek, site 7; (c) War Eagle Creek, site 2; (d) White River, site 10; —, seasonal base flow; …, total stream flow; ●, base flow water quality sample; ○, surface runoff water quality sample

the White River gauge site (where basin size was largest), to 1 m^3 s^{-1} at Brush Creek and 2–3 m^3 s^{-1} at Richland and War Eagle Creeks. However, maximum discharge for floods did increase in a downstream direction (*i.e.* with basin size), as the largest floods occurred in White River. Hydrograph separation suggested that water-quality samples were collected when seasonal base flow was less than 70% of total flow, *e.g.* surface runoff or storm events, in 8, 9, 10 and 18 out of 33 sampling dates in Brush Creek, Richland Creek, War Eagle Creek and the White River, respectively.

3.2. *Stream nutrient concentrations*

In these streams, N is far more plentiful than P, and N:P ratios are consistently 80–110:1. Inorganic N is dominated by NO_3, whereas NH_4 was generally below detection limits. At all sites average NO_3 and TN concentrations were greater in year 1 than year 2, but no consistent annual trend was evident for either SRP or TP. Seasonal patterns of nutrient concentrations varied annually and between sites; however, N and P concentrations generally peaked in concentration during both autumn and spring high flows.

Only the sub-basin outlets (sites 2, 3, 7, 8 and 9) were used to examine the relationship between land use and annual log-mean nutrient concentrations in the water column of the respective streams. In the White River sub-basin, sites 8 and 9 were used to avoid external influences on nutrient concentrations from the City of Fayetteville's wastewater treatment plant. Sites 2 and 7 were the basin outlets for War Eagle and Richland Creek watersheds. A positive correlation was observed between the proportion of pasture in the sub-basins

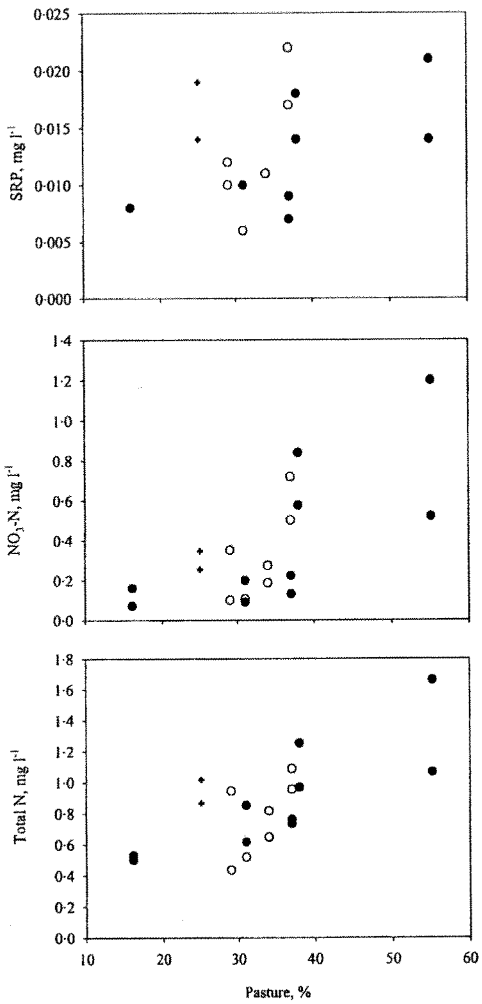

Fig. 3. Geometric mean soluble reactive phosphorus, NO_3-N, and total nitrogen as a function of per cent pasture in the Beaver Lake Basin; SRP, soluble reactive phosphorus; ●*, basin outlets, sites 2, 3, 7–9;* ○*, upstream sites, sites 1, 4–6; +, point source impacted site, site 10*

(arc-sin square root transformed, Zar, 1984) and annual geometric-mean SRP (regression coefficient $R = 0{\cdot}73$, number of samples $n = 10$, probability $P < 0{\cdot}02$), NO_3 ($R = 0{\cdot}76$, $n = 10$, $P < 0{\cdot}01$) and TN ($R = 0{\cdot}87$, $n = 10$, $P < 0{\cdot}001$) (*Fig. 3*), whereas NH_4, TKN and TP were not significantly correlated. Pasture land use accounted for over 50% of the variation of the change in annual geometric-mean SRP, NO_3-N and TN concentrations between sub-basins outlets.

3.3. *Stream nutrient export*

The relationships between nutrient concentrations [ln(*C*)] and discharge [ln(*Q*)] were variable between constituents and were not consistently significant; however, the relationship between nutrient loads [ln(*L*)] and discharge were consistently significant for all parameters ($P < 0{\cdot}05$). In some cases a significant trend in concentration and/or load with time and season was displayed, but results were variable within and among streams. The concentration model explained between 8 and 57% of the variation in nutrient concentrations, whereas the load model explained between 40 and 96% in nutrient loads.

Average N and P export from the White River watershed was greater than from any other sub-watershed (Table 2), but average unit-area N and P yield was least. Conversely, average annual N and P export in Brush Creek watershed was least but when weighted on a unit area basis the export was greatest. Annual nutrient export minus the wastewater treatment plant (WWTP) load (FTN Associates, Ltd, 1992) increased with increasing basin size in the four sub-watersheds of Beaver Lake, whereas unit area nutrient export decreased exponentially with watershed size (*Fig. 4*). Total N, NO_3 and SRP export per unit area increased exponentially with per cent pasture, whereas TP export per unit area was not related exponentially to per cent pasture in the sub-basins (*Fig. 4*).

4. Discussion

4.1. *Stream nutrient concentrations*

In Northwest Arkansas, the Beaver Lake/White River Basin contains over 2000 poultry houses in an area of approximately 300 000 ha producing about five flocks of poultry per year and 200 000 $Mg\,yr^{-1}$ of poultry litter. In general, poultry litter is approximately 4% TN and 2% TP (Moore *et al.*, 1997), representing a TN and TP source of about 8000 and 4000 $Mg\,yr^{-1}$, respectively, in the Beaver Lake Basin. Most often, poultry litter is land applied as a fertiliser to pastures; however, some poultry litter is used as alternative by-products such as feed supplements or compost. Approximately 80% of the poultry houses are contained with the subwatersheds where nutrient concentrations and export were measured.

Table 2
Estimated nutrient loading and the confidence interval (95% CI) from the regression models of the four gauged sub-basins in the Beaver Lake Basin

Sub-basin	*Yr*	*SRP, Mg*	*(95% CI)*	*TP, Mg*	*(95% CI)*	*NO_3-N, Mg*	*(95% CI)*	*TN, Mg*	*(95% CI)*
Brush	1	2·0	(1·2–3·3)	3·5	(2·2–5·4)	101	(51–180)	130	(86–187)
Creek	2	1·6	(0·9–2·8)	3·5	(2·0–5·7)	58	(30–105)	97	(63–142)
	avg	1·8		3·5		80		113	
Richland	1	3·0	(1·8–4·8)	9·1	(5·5–14·3)	201	(77–445)	278	(170–430)
Creek	2	5·0	(2·6–8·7)	16·9	(8·7–29·9)	158	(42–434)	361	(198–610)
	avg	4·0		13·0		180		320	
War Eagle	1	8·4	(4·4–14·8)	14·3	(9·4–20·8)	279	(172–430)	370	(251–527)
Creek	2	7·1	(3·4–13·4)	10·5	(6·5–16·1)	202	(115–330)	301	(194–448)
	avg	7·8		12·4		240		336	
White	1	15·5	(5·3–37·0)	43·2	(23·3–74·1)	363	(106–934)	530	(340–795)
River	2	25·4	(9·4–55·8)	49·0	(28·2–79·7)	362	(109–898)	592	(390–862)
	avg	20·4		46·1		362		561	
Total	1	29·0	(12·7–59·8)	70·1	(40·5–115)	943	(406–1990)	1307	(847–1939)
Export	2	39·1	(16·3–80·8)	79·9	(45·3–131)	781	(296–1767)	1351	(845–2063)
	avg	34·1		75·0		862		1329	

Yr, year; SRP, soluble reactive phosphorus; TP, total phosphorus; NO_3-N, nitrate nitrogen; TN, total nitrogen; avg, average load of years 1 and 2 ($Mg\,yr^{-1}$).

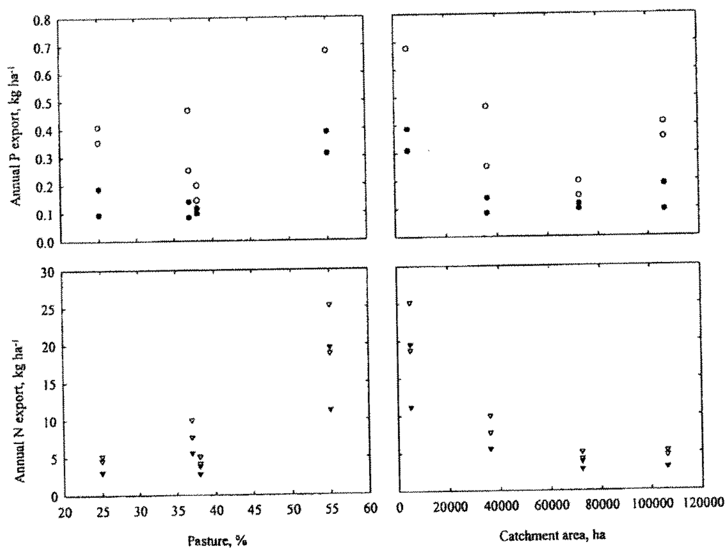

Fig. 4. Annual nutrient export as a function of per cent pasture and catchment area in the Beaver Lake Basin; ●, soluble reactive phosphorus; ○, total phosphorus; ▼, NO_3-N; ∇, total nitrogen

When P fertilisers are applied to pastures, the P concentrates in the top portion of the soil (Kingery *et al.*, 1994). Although P leaching can occur in deep sandy organic soils (Sims *et al.*, 1998), P does not leach in Northwest Arkansas soils because the silt loams of the region have a higher clay content with depth, and these higher clay sub-soils have increased P buffering capacity. Thus, the mechanism for P loading into the streams is via surface runoff, which varies in each sub-basin with soil type, slope, vegetation, antecedent moisture, management, *etc.* Pastures in northwest Arkansas have shown a positive correlation between soil P and P in runoff water

(Pote *et al.*, 1999). Furthermore, Edwards and Daniel (1993) reported an increasing relationship between P loading from poultry litter to soils and runoff P levels.

However, even though soil P levels or P loading onto a certain pasture may be high, inorganic and sediment-bound P in runoff can be low or even zero if there is no surface runoff. Loss of P from upland areas is regulated by P source factors (soil P, soil fertiliser and management) combined with P transport factors (runoff and erosion) (Gburek & Sharpley, 1998). The flow path of P transport from all positions throughout the basin has considerable effect on the amount of P reaching streams, rivers, and eventually reservoirs and lakes (Gburek & Pionke, 1995). The source of surface runoff may be variable within the watershed, and certain portions of the watershed may produce large amounts of surface runoff.

Unlike P, NO_3 molecules have low affinity for exchange or covalent bonding in soils and seldom form a precipitant. Whereas P transport is via overland flow, NO_3 movement may be through surface runoff, subsurface flow and/or groundwater flow. Nitrate in excess of plant requirements may leach through the soil and reach the stream via groundwater or inter-flow (Lowrance, 1992). Various studies have confirmed that agricultural leachates were a major source of NO_3 contamination in subsurface flow and ground water (Lowrance *et al.*, 1984; Sharpley *et al.*, 1987). The Beaver Lake Basin contains karstic geological features that allow for ground water movement without much natural filtering. The range of NO_3-N concentrations in streams draining the Beaver Lake Basin was from an annual geometric mean of 0·1 to 1·2 mg l^{-1}. These high concentrations are probably reflective of regional ground water enrichment associated with agricultural activity within the Ozark Plateau (Petersen *et al.*, 1999).

Regardless of the nature of the flow path with which terrestrial applied nutrients reach aquatic ecosystems, the results of this investigation suggest the proportion of pastureland in the sub-basins is an important determinant in annual stream nutrient concentrations. Increasing nutrient concentrations were observed, with increasing proportion of pasture in streams of the Beaver Lake Basin, and several investigations have shown similar increasing relationships in basins throughout the United States (Byron & Goldman, 1989; Jordan *et al.*, 1997; McFarland & Hauck, 1999). Thus, land application of poultry manure to pastures in northwest Arkansas may have impacted in-stream nutrient concentrations. Although per cent pasture did explain a significant portion of the variation (about 50%) in annual geometric mean nutrient concentrations in streams, many other factors such as nutrient cycling and processing in the upland area, riparian zone, surface–ground water interaction and the stream channel also contribute to the variability of stream nutrient concentrations (Fenn & Poth, 1999; Meyer *et al.*, 1988).

4.2. *Stream nutrient export*

In the Beaver Lake Basin, 65 and 45% of the P and N entering the reservoir is the form of SRP and NO_3, respectively. Thus, a large portion of the nutrient load is considered to be bioavailable to algae in the reservoir. Overall, the greatest amount of nutrients were transported from the White River watershed (the largest catchment) and the least from Brush Creek watershed (the smallest catchment). Prairie and Klaff (1986) suggested that stream TP export from basins dominated by pasture (greater than 90% on an area basis) was an exponential function of catchment size, but TP yield from forest systems was a linear function of catchment size. Nutrient yields from streams draining the Beaver Lake Basin decreased exponentially with catchment size despite exhibiting a slight dominance of forest in the sub-basins. However, nutrient yields in these sub-basins also increased exponentially with the proportion of pasture. Furthermore, the per cent pasture in the four sub-basin outlets was approximately a linear function of catchment size ($R = 0{\cdot}92$, $n = 4$, $P < 0{\cdot}08$) and none of the sub-basins were dominated (greater than 90%) by any one-land use category. Thus, the proportion of pasture in these sub-basins not only impacted absolute nutrient concentrations (annual geometric mean) but most likely nutrient export in the streams draining the basin.

The annual average TP export from the Beaver Lake Basin was 0·34 kg ha^{-1} yr^{-1}, within the range of export for both pastures (0·3 – 2·8) and forests (0·1 – 0·4) in the USA (Beaulac & Reckhow 1982, Young *et al.* 1996). Annual average TN export (6·0 kg ha^{-1} yr^{-1}) in this basin was greater than the range (2–3·5 kg ha^{-1} yr^{-1}) reported for forest systems in the USA, but well within the range of TN loss from streams draining catchments dominated by pastures (2–11 kg ha^{-1} yr^{-1}). Nutrient yields are also about half of that observed in the Illinois River Basin during 1997 through 1999, an Ozark Plateaus catchment near this basin (Green & Haggard, 2001). The nutrient yields from the Beaver Lake Basin were compared to the average nutrient yield in undeveloped, pristine streams during 1990 through 1995 (Clark *et al.*, 2000). Average annual NO_3 and TN yields were over 10 and 6 times greater, respectively, in the Beaver Lake Basin than the average values reported for all streams of the Hydrologic Benchmark Network, North Sylamore Creek in the Ozark Plateau (north-central Arkansas, USA) and the Cossatot River in the Ouchita Mountains (southeast Oklahoma and southwest Arkansas, USA) (*Fig. 5*). Similarly, SRP and TP export were over three times greater in the Beaver

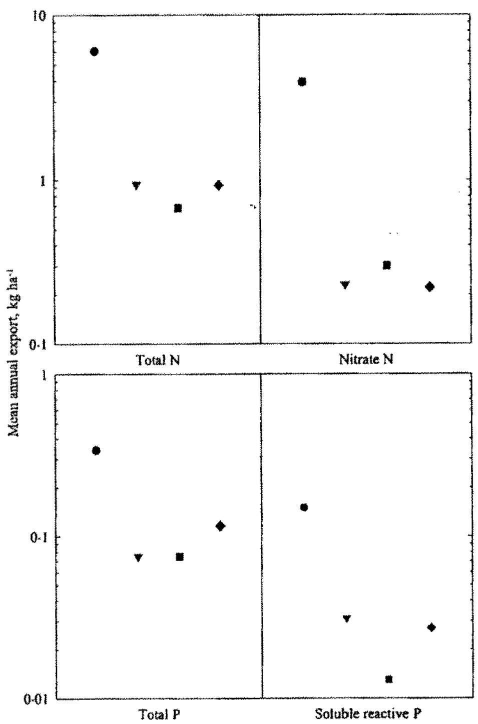

Fig. 5. Comparison between Beaver Lake Basin nutrient yields and those of relatively undeveloped areas reported by Clark et al. (2000); ●, average from this study, 1993–1995; ▼, average from undeveloped catchments, 1990–1995; ■, average from North Sylamore Creek, 1990–1995; ◆, average from Cossatot River, 1990–1995

Lake Basin. These differences are possibly reflective of the presence of confined animal agriculture and the practice of land applying animal manure because the area is not dominated by urban-suburban land use or row-crop agriculture (generally less than 4%).

The nutrient loads reported within this report were compared to the National Eutrophication Survey (EPA, 1977) and a comprehensive study by FTN Associates, Ltd (1992) (Table 3). It appears that TP and TN loads into this reservoir are equivalent to pre-tertiary WWTP treatment in the mid-1970s, and the TP and TN loads are approximately 1·5 and 1·3 times greater than reported in 1992. However, nutrient transport and loads within streams are dynamic and highly dependent on the amount of water passing a fixed point because load is a function of discharge. Thus, loads are not always comparable between years because of variation in annual runoff.

In the Beaver Lake Basin, the poultry industry produces a considerable amount of N and P (8000 and 4000 Mg yr^{-1}) in the Beaver Lake Basin; it is assumed that about 80% is produced in the subwatersheds where N and P export were measured. The N and P exported in the Beaver Lake subwatersheds was about 1290 and 70 Mg yr^{-1}, excluding WWTP inputs; this amount is only 16 and 2% of the total N and P produced from the poultry industry. Similar percentages have been estimated in neighbouring catchments (Storm *et al*., 2002; Nelson *et al*., 2002) with large numbers of poultry houses.

5. Conclusions

Ten stream sites were sampled from August 1993 through July 1995 17 times per year in the Beaver Lake Basin, southwestern portion of the Ozark Plateaus in northwest Arkansas, USA. Stream soluble reactive phorphorus (SRP), NO_3-N and total nitrogen (TN)

Table 3
Nutrient loading estimates from this study, the National Eutrophication Survey (EPA, 1977), and FTN Associates, Ltd (1992)

Sub-basin	*EPA (1977)*		*FTN (1992)*		*Current Study*[*]	
	TP, Mg yr^{-1}	*TN, Mg yr^{-1}*	*TP, Mg yr^{-1}*	*TN, Mg yr^{-1}*	*TP, Mg yr^{-1}*	*TN, Mg yr^{-1}*
White River	10·4	443	22·7	399	46·1[†]	560[†]
Richland Creek	3·6	113	3·8	98	13·0	320
War Eagle Creek	12·1	391	12·7	439	12·4	370
WWTP	43·5	155	6·2	39		
Total	69·5	1103	45·4	975	71·5	1250

TP, total phosphorus; TN, total nitrogen; WWTP, wastewater treatment plant; total, sum of export from the White River, Richland Creek and War Eagle Creek sub-basins and WWTP.
[*] average from both years in this study.
[†] In the current study White River loading includes WWTP.

concentrations (geometric-mean) increased linearly with per cent of pasture in each sub-watershed, whereas N and P export coefficients increased exponentially with pasture land use. Nutrient export in $kg\,yr^{-1}$ increased with basin size, but nutrient yield in ($kg\,km^{-2}\,yr^{-1}$) decreased with basin size. This investigation emphasised the need to carefully manage poultry litter because small losses of nutrients compared to the total amount of nutrients produced in a basin may still impact stream nutrient concentrations and export. Furthermore, nutrient export in this basin is considerably greater than that observed in relatively undeveloped basins across the USA, specifically in two catchments in Arkansas, USA. Nutrient uptake, transformation and transport from the terrestrial through the aquatic ecosystem must be addressed to fully understand the impacts of increased agricultural land use on stream nutrient concentrations and export.

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References

Alexander R B; Smith R A (1990). County-level estimates of nitrogen and phosphorus fertilizer use in the United States. Publication 90-130, US Geological Survey Open-File Report

APHA (1992). Standard Methods for Examining Water and Waste Water. (18th Edn.). American Public Health Association, Washington, DC

Arkansas Agricultural Statistics Service (1995). Arkansas Agricultural Statistics—1994. Arkansas Agricultural Experiment Station Report. Series. 325. University of Arkansas, Fayetteville

Beaulac M N; Reckhow K H (1982). An examination of land use and nutrient export relationship. Water Resources Bulletin, **18**, 1013–1024

Byron E R; Goldman C R (1989). Land-use and water quality in tributary streams of Lake Tahoe, California—Nevada. Journal of Environmental Quality, **18**, 84–88

Carpenter S R; Caraco N F; Correll D L; Howarth R W; Sharpley A N; Smith V H (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications, **8**, 559–568

Clark G M; Mueller D K; Mast M A (2000). Nutrient concentrations and yields in undeveloped stream basins of the United States. Journal of American Water Resources Association, **36**, 849–867

Cohn T A; DeLong L L; Gilroy E J; Hirsh R M; Wells D K (1989). Estimating constituent loads. Water Resources Research, **25**, 937–942

Davis J V; Bell R W (1998). Water quality assessment of the Ozark Plateaus Study Unit, Arkansas, Kansas, Missouri, and Oklahoma: Nutrients, bacteria, organic carbon, and suspended sediment in surface water, 1993–95. Publication 98-4164, US Geological Survey Water Resources Investigations Report

Edwards D R; Daniel T C (1993). Effects of poultry litter application rate and rainfall intensity on quality of runoff from fescue plots. Journal of Environmental Quality, **22**, 361–365

EGIS (1996). Beaver Lake, Arkansas, Water Quality Enhancing Project Water Quality Monitoring Study. US Army Corps of Engineers and Arkansas Soil and Water Commission

EPA (1977). National Eutrophication Survey. Report on Beaver, Table Rock, and Bull Shoals Reservoirs, Arkansas, and Taneycomo Reservoir, Missouri. Working Paper #480, US Environmental Protection Agency Region VI and VIII

EPA (1983). Methods for chemical analysis of water and wastes. US Environmental Protection Agency Report 600/4-79-020, US Government Printing Office, Washington, DC

Fenn M E; Poth M A (1999). Temporal and spatial trends in stream water nitrate concentrations in the San Bernardino Mountains, Southern California. Journal of Environmental Quality, **28**, 822–36

Ferguson R I (1986). River loads underestimated by rating curves. Water Resources Research, **22**, 74–76

FTN Associates, Ltd (1992). Final Report for Beaver Lake. Phase I. Diagnostic/Feasibility Study. Vol. 1, Book A. Prepared for Arkansas Department of Pollution Control and Ecology

Gburek W J; Pionke H B (1995). Management strategies for land-based disposal of animal wastes: hydrologic implications. In: Animal Waste and the Land–Water Interface (Steele K, ed.), pp 313–324, Lewis Publishers, New York

Gburek W J; Sharpley A N (1998). Hydrologic controls on phosphorus loss from upland an agricultural watershed. Journal of Environmental Quality, **27**, 267–277

Green W R; Haggard B E (2001). Nitrogen and phosphorus concentrations and loads in the Illinois River, Arkansas, 1997–1999. Publication 01-4217, US Geological Survey Water Resources Investigation Report

Institute of Hydrology (1980). Low flow studies. Rep. No. 1, 21pp, Wallingford, Oxon, United Kingdom

Jordan T E; Correll D L; Weller D E (1997). Effects of agriculture on discharge of nutrients from coastal plain watersheds of Chesapeake Bay. Journal of Environmental Quality, **26**, 836–848

Kingery W L; Wood C W; Delaney D P; Williams J C; Mullins G L (1994). Impact of long-term land application of broiler litter on environmentally related soil properties. Journal of Environmental Quality, **23**, 139

Lowrance R (1992). Groundwater nitrate and denitrification in a coastal plain riparian forest. Journal of Environmental Quality, **21**, 401-405

Lowrance R; Todd R; Fail Jr. J; Hendrickson Jr O; Leonard R; Asmussen L (1984). Riparian forests as nutrient filters in agricultural watersheds. Bioscience, **35**, 374–7

McFarland A M S; Hauck L M (1999). Relating agricultural land uses to in-stream storm water quality. Journal of Environmental Quality, **28**, 836–844

Meyer J L; Mc Dowell W H; Bott T L; Elwood J W; Ishizaki C;Melack J M; Peckarsky B L; Peterson B J; Rublee P A (1988). Elemental dynamics in streams. Journal of North American Benthological Society, **7**, 410:432

Moore Jr P A; Haggard B E; Daniel T C; Edwards D R; Shreve B R; Sauer T J (1997). Demonstration of nutrient management for poultry litter using alum precipitation of soluble phosphorus. Final Report to Region VI US Environmental Protection Agency for Task 92-500

National Research Council (1992). Restoration of Aquatic Ecosystems. National Academy Press, Washington, DC

Nelson M A; Soerens T S; White K L (2002). Water-quality sampling and mass balance in the Illinois River. Proceedings Report Arkansas Water Resources Center Annual Conference, Fayetteville, Arkansas, 23-24 April, 2002

Newman A (1996). Water pollution point sources still significant in urban areas. Environmental Science and Technology, **29**, 114

Petersen J C (1992). Trends in stream water quality data in Arkansas during several time periods between 1975 and 1989. US Geological Survey Water-Resources Investigation Report, Publication 92-4044

Petersen J C; Adamski J C; Bell R W; Davis J V; Femmer S R; Freiwald D A; Joseph R L (1999). Water Quality in the Ozark Plateaus, Arkansas, Kansas, Missouri, and Oklahoma, 1992-95. US Geological Survey Circular, Publication 1158

Pote D H; Daniel T C; Nichols D J; Sharpley A N; Moore P A Jr.; Miller D M; Edwards D R (1999). Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff. Journal of Environmental Quality, **28**, 170–175

Prairie Y T; Klaff J (1986). Effect of catchment size on phosphorus export. Water Resources Bulletin, **22**, 465–470

Robertson D M; Roerish E D (1999). Influence of various water quality sampling strategies on load estimates for small streams. Water Resources Research, **35**, 3747–3759

Sharpley A N; Smith S J; Naney J W (1987). Environmental impact of agricultural nitrogen and phosphorus use. Journal of Agricultural and Food Chemistry, **35**, 812-817

Sims G K; Ellsworth T R; Mulvaney R L (1995). Microscale determination of inorganic nitrogen in water and soil extracts. Communications in Soil Science and Plant Analysis, **26**, 303–316

Sims J T; Simard R; Joern B (1998). Phosphorus loss in agricultural drainage: historical perspective and current research. Journal of Environmental Quality, **27**, 277–292

Storm D E; White M; Smolen M D (2002). Modeling the Lake Eucha Basin using SWAT 2000. Final Report submitted to the Tulsa Metropolitan Utility Authority, Department of Biosystems and Agricultural Engineering, Oklahoma State University, Stillwater, OK, 9 August, 2002

Technicon (1976). Individual/simultaneous Determination of Nitrogen and/or Phosphorus in BD Acid Digests. Industrial Method No. 329-74W/A. Technicon Industry Sytems, Tarrytown, NJ

Young W J; Marston F M; Davis J R (1996). Nutrient exports and land use in Australian Catchments. Journal of Environmental Management, **47**, 165–183

Wahl K L; Wahl T L (1995). Determining the flow of Comal Springs at New Braunfels, Texas. In: Texas Water '95, pp 77-86. American Society of Civil Engineers, 6-17 August, 1995, San Antonio, TX

Zar J H (1984). Biostatistical Analysis. (2nd Edn.). Prentice-Hall, Englewood, NJ